

SPATIAL AND TEMPORAL VARIATION OF BEACHED BIRDS IN NORTHERN ARGENTINE PATAGONIA

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Received 27 June 2024, accepted 13 September 2024

ABSTRACT

Pizá, V. M., Pereyra, P. J., & Seco Pon, J. P. (2025). Spatial and temporal variation of beached birds in northern Argentine Patagonia. *Marine Ornithology*, 53(1), 99–110. <http://doi.org/10.5038/2074-1235.53.1.1620>

Beached birds on marine coastal beaches, along with other megafauna, are a valuable source of biological information, offering insights into species occurrence and factors that negatively affect their populations. The main objective of this study was to evaluate the spatial and temporal variation, and species composition, of coastal and marine birds found on sandy beaches in the Bahía de San Antonio Protected Natural Area (Río Negro Province, Argentina), and to relate resulting species abundance and richness to oceanographic and environmental variables. A biweekly survey was carried out over two years (2020–2022), during which 672 beached birds were recorded. The overall encounter rate of beached birds was 1.35 birds/km. Most birds were identified to the species level, totaling 27 species belonging to 11 orders. The three most frequently encountered species were Magellanic Penguin *Spheniscus magellanicus* (49.38%), Kelp Gull *Larus dominicanus* (43.75%), and Neotropic Cormorant *Nannopterum brasilianum* (22.5%). The height of high tide and wind direction significantly affected beached bird abundance at various spatial and temporal scales. We conclude that the assemblage of beached birds is heterogeneous, mainly consisting of species that inhabit the Río Negro Province coast, rather than those from farther offshore. Future studies in the area should include drift experiments and cover a larger geographic scale.

Key words: Argentina, coastal and marine birds, Kelp Gull, Magellanic Penguin, Neotropic Cormorant, beached birds

INTRODUCTION

Coastal and marine birds are often considered reliable indicators of the health of marine ecosystems (Burger & Gochfeld, 2004; Mallory et al., 2010), and studies examining marine bird carcasses found on beaches—used as a proxy for the living community—have increased since the 1970s (e.g., Diamond et al., 2020; Powlesland, 1986; Veitch, 1978). Due to their particular life history traits, including high trophic position and longevity, birds accumulate exposure to anthropic stressors over large geographic areas. Thus, they are well-suited to serve as indicators of levels of chemical pollution, including oil spills (Camphuysen & Heubeck, 2001; Furness & Camphuysen, 1997), organic pollutants (Malcom et al., 2003; Sagerup et al., 2014), harmful algal blooms (Ben-Gigirey et al., 2021; Shumway et al., 2003), and marine debris, such as abandoned, lost, or discarded fishing gear (Kühn & Van Franeker, 2020; Phillips et al., 2010; Roman et al., 2019). Currently, a third of the 346 extant seabird species face some type of threat, and their global populations have significantly declined from 1950 to 2010 (Croxall et al., 2012; Grémillet et al., 2018; Paleczny et al., 2015). Therefore, monitoring bird populations—or even their proxies, such as beached birds—is paramount for developing effective biological management and conservation strategies (Mace & Baillie, 2007; Thomas, 1996).

Coastal and marine birds (along with other coastal wildlife) are considered beached when found dead, either on the beach or floating in the water, or when alive on the beach and unable to

return to the water (Geraci & Lounsbury, 1993, 2005). Long-term systematic surveys of beached birds are an accessible, cost-effective, and easy-to-implement tool for obtaining information (i.e., species composition), and can also be used to determine the spatial and temporal patterns of species occurrence and their relative abundance in a certain area (Byrd et al., 2014; Hamel et al., 2009). These surveys can also provide insights into the interaction between birds and human activities (Nevins et al., 2011; Simeone et al., 2021; Zydelis et al., 2013). Surveys of beaches to detect beached birds have occurred since the middle of the 20th century along the coasts around the world (e.g., Camphuysen & Heubeck, 2001; Heubeck, 1995; Powlesland & Imber, 1988). The primary objective of this type of survey was (and remains) to establish patterns of beached bird occurrence and identify the factors contributing to their presence. The results can reveal long-term changes in the distribution and abundance of various species (and other megafauna), with a particular emphasis on oil-contaminated animals (e.g., Camphuysen & Vollaard, 2016; Larsen et al., 2007; Stowe & Underwood, 1984).

While beached bird surveys occur regularly around the globe (Bodkin & Jameson, 1991; Haman et al., 2013; Roletto et al., 2003), in the Southern Hemisphere there is a spatial bias towards coastal areas of Australia and New Zealand (Powlesland, 1986; Taylor, 2004), Chile (Simeone et al., 2021), and Perú (Garate, 2013; Ortiz-Alvarez et al., 2022). In the South Atlantic, studies have focused on South Africa (Avery, 1984; Batchelor, 1981) and Brazil (Mariani et al., 2019; Martuscelli et al., 1997). In Argentina, surveys have been centered

in the Buenos Aires Province (Korschenewski, 1975; Narosky & Fiameni, 1986; Seco Pon & García, 2022), and to a lesser extent Tierra del Fuego, the Antarctic, and the South Atlantic Islands Province.

In northern Argentine Patagonia, the coastal biomes of the Río Negro Province are considered valuable breeding and feeding habitats for a great variety of birds, including coastal species (González, 2007; Llanos et al., 2011). In fact, several coastal areas are listed as Important Bird Areas (IBAs), including El Cóndor and San Antonio Oeste (Di Giacomo & Coconier, 2007). However, limited information is available along the Río Negro Province coasts despite the presence of valuable protected areas. For instance, the San Antonio Oeste IBA lies close to Bahía de San Antonio Protected Natural Area. Given this context, the main goal of this study was to evaluate the spatial and temporal variation of beached coastal and marine birds in the Bahía de San Antonio Protected Natural Area, and to relate findings (abundance and richness) with oceanographic and environmental variables.

METHODS

Study area

The present study was carried out at the Bahía de San Antonio Protected Natural Area (hereinafter referred as to BSAPNA), Río Negro Province, northern Argentine Patagonia (40°47'00"S, 065°03'00"W). This area consists of both terrestrial (203 km²) and marine domains (609 km²). The terrestrial domain spans both governmental and private lands, encompassing the Port of San Antonio Este (381 inhabitants, 734 km²), San Antonio Oeste (16,265 inhabitants; 9,900 km²), and the village of Las Grutas (4,807 inhabitants; 8,151 km²) (Instituto Nacional de Estadística y Censos [INDEC], 2010). The governmental areas are fully managed by the Río Negro Province, with moderate enforcement of management policies. The presence of a permanent human population within boundaries of the BSAPNA, along with the coexistence of conservation actions and economic activities—such as fisheries, agriculture, livestock, and industry—complicates the overall management of the BSAPNA (Morea, 2019).

The BSAPNA is located in the Monte Ecoregion, characterized by plains and plateaus (Cabrera, 1976), and the Argentine Province North Patagonian Gulfs Ecoregion (Balech & Ehrlich, 2008), an area which includes sand beaches, shells and sandbars, and extensive marsh, as well as sandy plains furrowed by drainage channels that are exposed at low tide. The prevailing climate is temperate semi-arid (Thornthwaite, 1948), shaped by local variations due to the proximity of the sea. Average surface water temperature in the region is around 15.1 °C (Saad et al., 2019). During March to November (austral autumn to spring), winds from the northwest predominate, while in the summer, the dominant winds are from the southeast (Genchi et al., 2010). Local tides are semidiurnal, ranging from 6–9 m in amplitude. The associated currents are in the order of 1–2 m/s (Mazio & Vara, 1983). The most intense waves generally come from the southeast, reaching periods of 7–10 s and heights of 0.5–1.5 m (exceeding 3 m in storm events; Scalise et al., 2009; Kokot et al., 2013).

Fish diversity is relatively high in the BSAPNA—including both benthonic and pelagic fishes—many of which are regularly targeted for sport and artisanal fishing activities (De la Barra et al., 2019; González et al., 2010; Narvarte et al., 2007; Narvarte et al., 2011). The area also supports several important resting,

foraging, and breeding sites that are regularly used by several avian species (Blanco & Canevari, 1995; Giaccardi & Reyes, 2012; González, 2007).

Sampling

Within the BSAPNA, surveys were carried out in two sites known locally as (1) “Mar Grande” (40°45'22"S, 064°56'31"W) and (2) “Oasis” (40°46'19"S, 065°02'44"W), both sandy beaches located 8 km apart. The choice of the locations was based on their accessibility and/or by previous reports indicating the occurrence of beached marine vertebrates (Romero et al., 2021; Savigny & Carbajal, 2015; Fig. 1).

The data used in this study were collected through biweekly surveys at each sampling site. At each site, two adjacent transects, each approximately four kilometers long, were sampled. These transects ran parallel to the coastline, covering the entire beach. One transect encompassed the upper forebeach, and the other included the lower forebeach, with the location of the previous high tide serving as the reference point. For analysis, the upper and lower forebeaches were treated as separate levels. Each beach was surveyed independently and biweekly (two surveys per month) from June 2020 to June 2022 (24 months). Surveys were conducted on foot, as current legislation prohibits the use of engine-powered vehicles in this coastal area. For statistical analyses, the time periods from June 2020 to June 2021 and from June 2021 to June 2022 are referred to hereafter as the 2020/21 and 2021/22 periods, respectively.

Beached birds were registered in each transect. Each transect was covered once and all the sighted beached birds were pooled together. In this study, each transect represented a sampling unit, with a projected sample size of 192 transects for the beached bird transect survey. However, due to logistical challenges and COVID-19 restrictions, only 160 transects were completed. Beached birds encountered on each transect were photographed and identified to the lowest possible taxonomic level, either *in situ* or in the laboratory, using specialized guides (Narosky & Yzurieta, 2010; Povedano, 2016; Savigny, 2021). Depending on the degree of carcass preservation, each bird specimen was scored from 1 to 6 (minimum to maximum state of decay; adapted from Geraci & Lounsbury, 2005). The scores included the following values: “1” = live animal with external wounds, signs of malnutrition and/or dehydration; “2” = recently dead animal, fresh, odorless, *rigor mortis*; “3” = mild decomposition, internal organs intact, little odor, no eyes; “4” = advanced decomposition, detachment of feathers, skin and/or scales, swollen body due to the accumulation of gases, alteration of the color, texture and/or structure of the internal organs making their recognition difficult, strong odor; “5” = extremely advanced decomposition, dried body, presence of post-mortem wounds due to autolysis, parts of the skeleton visible, internal organs unrecognizable; and “6” = bones with or without remains of dry integument. Moreover, each seabird specimen was examined to determine the possible cause of death due to natural (physical condition, predation, parasite load) or anthropogenic (linear cuts, bullets wounds, entanglements in fishing gear, oil pollution, among others) causes. Carcasses were marked with colored wool and buried *in situ* to avoid double registration in successive outings.

Oceanographic and environmental data

Four days prior to each survey, data on oceanographic and environmental variables for the study area were collected; the

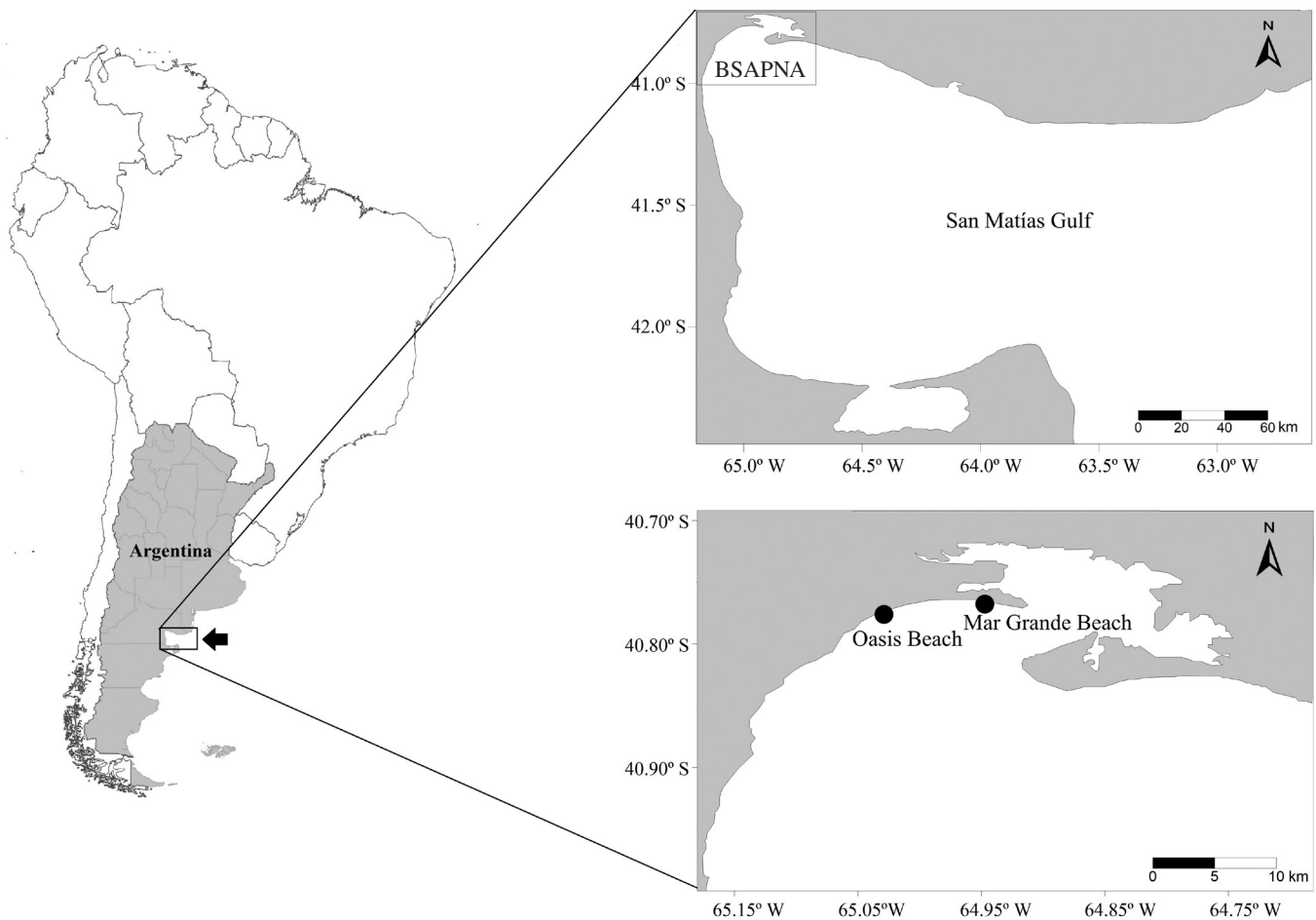


Fig. 1. Location of the study area in relation to Bahía de San Antonio Protected Natural Area (BSAPNA) in San Matías Gulf, Río Negro Province, Argentina.

variables that were chosen have been recorded in other beach surveys (Haman et al., 2013; Wiese & Elmslie, 2006). The height of high tides (m), and the intensity (knots) and direction (in degrees) of the tidal currents, were retrieved from the Naval Hydrography Service (Naval Hydrography Service, n.d.). Wave height (m), wind intensity (knots) and direction (degrees), and sea surface temperature (in °C) were retrieved from Windguru (Windguru, n.d.). A daily average value for each variable was calculated (without transformation). This approach—which considered variability in oceanographic and environmental conditions for the four days prior to the survey—assumed that if a coastal or marine bird died in the open sea, its carcass could be transported to the beach within that time frame (Brusius et al., 2020; Brusius et al., 2021; Vassallo, 2021).

Data analysis

The beached birds for each transect were characterized by abundance (number of specimens), occurrence (presence/absence), and species richness (S , number of species). The frequency of occurrence was defined as the percentage of transects in which each species or order was registered per survey, expressed as absolute (number of events) and relative (percentage) frequency of occurrence. Bird abundance was defined as the total number of each species or order found on the beach tallied during each transect. This metric was similar between sampling sites (Mar Grande: $n = 320$; Oasis: $n = 352$). Thus, no significant differences were found in the number of

beached birds across sampling sites (Kruskal-Wallis $H_{1;160} = 0.49$, $P = .48$). Therefore, the decision was made to combine both sampling sites for further analysis.

Differences in avian assemblages tallied between the lower and upper levels of each beach, and between seasons (intra- and inter-annual) of the sampled period, were analyzed separately using multivariate analysis techniques via the software package “PRIMER,” version 6.1 (Clarke & Gorley 2006). For this analysis, we applied hierarchical and multidimensional clustering of raw data (without transformation) using the Bray-Curtis similarity index. Analysis of similarities (ANOSIM) uses the Bray-Curtis similarity matrix to compute the R statistic, which varies between -1 and 1 , reaching its maximum value when all between-group dissimilarities are greater than all within-group dissimilarities. Statistical significance was determined by comparing the sample R with those produced by randomly assigning samples to groups (Clarke & Warwick, 2001). The P value of the test was calculated using the proportion of random arrangements with R values greater than the sample value. To test whether avian assemblages’ composition differed between levels of the beach within each sampled season (controlling for sampling period), we performed a two-way nested ANOSIM. Similarity percentages (SIMPER) were employed to determine the species that contributed the most to the dissimilarities between groups (Clarke, 1993; Clarke & Warwick, 2001). Only those species that contributed at least 10% of the similarity (see Results) were considered in the analysis.

To determine possible spatial or temporal variations in the number of individuals of the most abundant beached species, non-parametric analysis was conducted, since lack of normality in the data prevented us from conducting parametric analysis. When significant differences were detected, post hoc comparisons were carried out by pairwise comparisons. Pairwise comparisons were tested using a Wilcoxon rank-sum test (Wilcoxon, 1945).

The effect of oceanographic, environmental, spatial, and temporal variables on the overall abundance of beached birds was evaluated using generalized linear models (GLMs), with a negative binomial distribution. For species richness, the same GLM approach was used, employing a Poisson distribution (Crawley, 2007; Zuur et al., 2009). Potential correlations among oceanographic (height of high tide, intensity and direction of tidal current, wave height, and sea surface temperature) and environmental (intensity and wind direction) variables were identified *a priori* by estimating all pairwise Spearman rank correlation coefficients. Following this analysis, the oceanographic variables retained for the GLM model were height of high tide and wave height, while the environmental variable was wind direction (categorical variable: north, south, east, and west quadrant). Spatial and temporal variables were level of the beach (categorical variable: lower and upper) and season of sampled period (categorical variable per period: autumn, winter, spring, and summer), respectively. The fitness of the models was checked in DHARMA diagnostic plots using the “DHARMA” package (Hartig, 2022).

Total encounter rate was calculated by combining all species (including those unidentified) and relating the total number of beached birds to the beach linear distance covered during the survey. The same method was applied to the most abundant species. All analyses, except multivariate, were performed using R, version 4.1.2 (R Core Team, 2021). In all cases, differences were considered significant when $P < .05$. All reported values are the means (\pm 2 standard deviations [SD]).

RESULTS

Composition of beached bird assemblages

A total of 672 beached birds were recorded, with a maximum of 30 individuals per transect (Table 1). All birds were found dead. A high percentage of the birds (98%, $n = 657$) were identified to species, representing 27 species across 11 orders. A small fraction (2%, $n = 15$) could not be identified. Birds associated with coastal habitats (87%, $n = 569$) were more abundant than those associated with pelagic habitats (13%, $n = 88$; Table 1).

The most abundant and frequent beached species were Magellanic Penguin *Spheniscus magellanicus* (49.38%, $n = 250$), followed by Kelp Gull *Larus dominicanus* (43.75%, $n = 123$) and Neotropic Cormorant *Nannopterum brasilianum* (22.5%, $n = 59$; Table 1). An average total encounter rate of 1.35 ± 1.59 birds was estimated per linear kilometer (birds/km). Among these three species, the Magellanic Penguin encounter rate was 0.50 ± 0.84 birds/km, and the encounter rate for Kelp Gulls and Neotropic Cormorants was 0.25 ± 0.39 and 0.11 ± 0.28 birds/km, respectively.

Beached bird assemblages across temporal and spatial scales

Regardless of the temporal scale, the overall percentage of similarity among beached bird assemblages was comparable

between beach levels (lower forebeach = 12%, upper forebeach = 25%). In the lower forebeach, Magellanic Penguins accounted for 53.2% of the average similarity, followed by Kelp Gulls (28.8%). In the upper forebeach, these species made comparable contributions to the overall similarity (Magellanic Penguins 46.8%, Kelp Gulls 29.5%), with Neotropic Cormorants contributing 10.2% (Table A1 in Appendix, available on the website). The composition of species assemblages varied significantly between levels of the beach (two-way ANOSIM, $P = .002$; Fig. A1 in Appendix). Differences were mainly driven by the relative contribution of Magellanic Penguins (30%), Kelp Gulls (20.4%), and Neotropic Cormorants (11.3%) (Table A2 in Appendix). Other species contributed < 10% of the total species composition. Beached bird richness was significantly higher in the upper forebeach (Kruskal-Wallis $H_{1,160} = 44.11$, $P < .001$), with 33 species recorded, compared to only 23 species recorded in the lower forebeach (Table A3 in Appendix).

Irrespective of the spatial scale, the overall percentage of similarity among beached bird assemblages was relatively analogous among seasons (average similarity value for the 2020/21 and 2021/22 periods: autumn = 16.6%, winter = 10.7%, spring = 15.6%, summer = 26.8%). In autumn (both sampled periods combined), Magellanic Penguins contributed 34.5% of the average similarity, followed by Kelp Gulls (19.2%). In winter, Magellanic Penguins contributed 31.1% of the average similarity, followed by unidentified gulls (14.5%), Kelp Gulls (13.6%), and the Great Grebe *Podiceps major* (10.2%). In spring, Kelp Gulls contributed 28.2% of the average similarity, followed by Magellanic Penguins (23.6%) and Neotropic Cormorants (22.9%). Finally, in summer, Magellanic Penguins contributed 39.5% of the average similarity, followed by Kelp Gulls (11.9%), Manx Shearwaters *Puffinus puffinus* (10.3%), and unidentified gulls *Larus* spp. (10.2%) (Table A4 in Appendix). The composition of beached bird assemblages varied significantly between seasons (two-way ANOSIM $P = .015$; Fig. A2 in Appendix).

During 2020/21, the lowest number of beached species was recorded in spring ($S = 12$), while the highest occurred in winter ($S = 21$). In 2021/22, the minimum number of beached species was again observed in spring ($S = 9$), but the maximum occurred in autumn ($S = 15$; Table A5 in Appendix). Two species (Magellanic Penguin, Kelp Gull) were found in all seasons, but other species were found only occasionally (Tables A6, A7 in Appendix). The number of beached birds did not vary significantly with respect to season during 2020/21 (Kruskal-Wallis $H_{3,82} = 4.76$, $P = .19$), nor during 2021/22 (Kruskal-Wallis $H_{3,78} = 7.06$, $P = .06$). This was also true during the entire study period (Kruskal-Wallis $H_{7,160} = 14.02$, $P = .05$). The two-way nested ANOSIM analysis indicated statistically significant differences in beached bird assemblages between levels of the beach when controlling for the sampled period (two-way ANOSIM, $P = .014$). Differences were mainly driven by the relative contribution of Magellanic Penguins (30.1%), Kelp Gulls (20.4%), and Neotropic Cormorants (11.3%) (Table A8 in Appendix).

When considering the most abundant beached species, both Magellanic Penguins (Kruskal-Wallis $H_{7,160} = 24.57$, $P = .0009$) and Neotropic Cormorants (Kruskal-Wallis $H_{7,160} = 19.23$, $P = .007$) showed significant differences in the quantities of beached individuals between seasons (Tables A9, A10 in Appendix).

TABLE 1
Bird species found beached throughout the study period ($n = 160$ transects)^a

Scientific and common names	<i>n</i>	% <i>n</i>	Mean \pm standard deviation	Max	%AF	%RF	National status (ARG)	Status (IUCN)
Sphenisciformes	250	37.20	1.56 \pm 2.61	18	79	49.38		
Magellanic Penguin <i>Spheniscus magellanicus</i> ^{b,c}	250	37.20	1.56 \pm 2.61	18	79	49.38	VU	LC
Charadriiformes	206	30.65	1.29 \pm 1.63	8	92	57.5		
Kelp Gull <i>Larus dominicanus</i> ^{b,c}	123	18.30	0.77 \pm 1.19	6	70	43.75	NA	LC
<i>Larus</i> spp.	52	7.74	0.33 \pm 0.71	4	37	23.13		
Brown-hooded Gull <i>Chroicocephalus maculipennis</i> ^{b,c}	3	0.45	0.02 \pm 0.14	1	3	1.88	NA	LC
South American Tern <i>Sterna hirundinacea</i> ^{b,c}	5	0.74	0.03 \pm 0.17	1	5	3.13	NA	LC
<i>Sterna</i> spp.	14	2.08	0.09 \pm 0.30	2	13	8.13		
Sandwich Tern <i>Thalasseus sandvicensis</i> ^b	1	0.15	0.01 \pm 0.08	1	1	0.63	NA	LC
<i>Stercorarius</i> spp.	1	0.15	0.01 \pm 0.08	1	1	0.63		
American Oystercatcher <i>Haematopus palliatus</i> ^{b,c}	3	0.45	0.02 \pm 0.14	1	3	1.88	NA	LC
Magellanic Oystercatcher <i>Haematopus leucopodus</i> ^b	1	0.15	0.01 \pm 0.08	1	1	0.63	NA	LC
Two-banded Plover <i>Anarhynchus falklandicus</i> ^{b,c}	2	0.30	0.01 \pm 0.11	1	2	1.25	NA	LC
White-rumped Sandpiper <i>Calidris fuscicollis</i>	1	0.15	0.01 \pm 0.08	1	1	0.63	NA	LC
Procellariiformes	87	12.95	0.54 \pm 1.53	13	38	23.75		
Southern Fulmar <i>Fulmarus glacialis</i>	8	1.19	0.05 \pm 0.33	3	4	2.50	NA	LC
Manx Shearwater <i>Puffinus puffinus</i>	27	4.02	0.17 \pm 0.84	8	13	8.13	NA	LC
<i>Puffinus</i> spp.	9	1.34	0.06 \pm 0.26	2	8	5.00		
Great Shearwater <i>Ardenna gravis</i>	16	2.38	0.10 \pm 0.42	3	11	6.88	NA	LC
<i>Ardenna</i> spp.	11	1.64	0.07 \pm 0.28	2	10	6.25		
White-chinned Petrel <i>Procellaria aequinoctialis</i>	1	0.15	0.01 \pm 0.08	1	1	0.63	A	VU
Southern Giant Petrel <i>Macronectes giganteus</i>	1	0.15	0.01 \pm 0.08	1	1	0.63	VU	LC
<i>Macronectes</i> spp.	2	0.30	0.01 \pm 0.11	1	2	1.25		
<i>Thalassarche</i> spp.	12	1.79	0.08 \pm 0.36	3	8	5.00		
Suliformes	76	11.31	0.48 \pm 0.96	7	48	30.00		
Neotropic Cormorant <i>Nannopterum brasilianum</i> ^b	59	8.78	0.37 \pm 0.92	7	36	22.50	NA	LC
Imperial Shag <i>Leucocarbo atriceps</i>	12	1.79	0.08 \pm 0.29	2	11	6.88	NA	LC
<i>Phalacrocorax</i> spp.	5	0.74	0.03 \pm 0.21	2	4	2.50		
Podicipediformes	21	3.13	0.13 \pm 0.45	3	15	9.38		
Great Grebe <i>Podiceps major</i> ^b	18	2.68	0.11 \pm 0.39	2	14	8.75	NA	LC
Silvery Grebe <i>Podiceps occipitalis</i> ^b	2	0.30	0.01 \pm 0.11	1	2	1.25	NA	LC
White-tufted Grebe <i>Rollandia rolland</i> ^b	1	0.15	0.01 \pm 0.08	1	1	0.63	NA	LC
Anseriformes	6	0.89	0.04 \pm 0.22	2	5	3.13		
Coscoroba Swan <i>Coscoroba coscoroba</i> ^b	2	0.30	0.01 \pm 0.16	2	1	0.63	NA	LC
Black-necked Swan <i>Cygnus melancoryphus</i> ^b	2	0.30	0.01 \pm 0.11	1	2	1.25	NA	LC
Red Shoveler <i>Spatula platalea</i> ^b	1	0.15	0.01 \pm 0.08	1	1	0.63	NA	LC
<i>Anas</i> spp.	1	0.15	0.01 \pm 0.08	1	1	0.63		
Phoenicopteriformes	4	0.60	0.03 \pm 0.16	1	4	2.50		
Chilean Flamingo <i>Phoenicopterus chilensis</i>	4	0.60	0.03 \pm 0.16	1	4	2.50	VU	NT
Pelecaniformes	2	0.30	0.01 \pm 0.11	1	2	1.25		
Black-crowned Night Heron <i>Nycticorax nycticorax</i> ^{b,c}	1	0.15	0.01 \pm 0.08	1	1	0.63	NA	LC
Heron, No ID (unidentified)	1	0.15	0.01 \pm 0.08	1	1	0.63		
Gruiformes	1	0.15	0.01 \pm 0.08	1	1	0.63		
<i>Fulica</i> spp.	1	0.15	0.01 \pm 0.08	1	1	0.63		
Falconiformes	3	0.45	0.02 \pm 0.14	1	4	2.50		
Peregrine Falcon <i>Falco peregrinus</i> ^{b,c}	1	0.15	0.01 \pm 0.08	1	1	0.63	NA	LC
Chimango Caracara <i>Milvago chimango</i> ^{b,c}	2	0.30	0.01 \pm 0.11	1	2	1.25	NA	LC
Strigiformes	1	0.15	0.01 \pm 0.08	1	1	0.63		
Burrowing Owl <i>Athene cunicularia</i> ^{b,c}	1	0.15	0.01 \pm 0.08	1	1	0.63	NA	LC
No ID (unidentified)	15	2.23	0.09 \pm 0.29	1	14	8.75		
Total beached birds	672	100.00						

^a AF = frequency of occurrence as presence/absence per transect; RF = percentage of occurrence where each order and species was present per transect throughout the study period; National Status Conservation of Argentina (ARG) = Categorization of the Ministerio de Ambiente y Desarrollo Sustentable & Aves Argentinas 2017 (VU = vulnerable, NA = not threatened, A = threatened); Conservation status according to the International Union for Conservation of Nature (LC = least concern, VU = vulnerable, NT = almost threatened). For all species, the minimum number of individuals recorded was 0.

^b Species that nests in the province of Río Negro.

^c Species that nests in the Bahía de San Antonio Protected Natural Area (BSAPNA). No ID = unidentified.

Preservation status of the bird carcasses

Most birds were found in a state of extremely advanced decomposition (code “5”, 42%, $n = 283$), followed by carcasses composed of body parts, bones with or without remains of dry integument (code “6”, 33%, $n = 221$), and birds found in an advanced decomposition state (code “4”, 15%, $n = 98$). To a lesser extent, recently dead birds (code “2”, 6%, $n = 40$), or birds in slight decomposition, were found, with intact internal organs, little odor, and absence of eyes (code “3”, 4%, $n = 30$) (Fig. 2).

Oceanographic, environmental, temporal, and beach-scale analysis

The global GLM model was used to determine the effect of oceanographic and environmental variability, along temporal and beach scales, on the overall abundance of beached birds. The model showed a good fit, with residuals following a normal distribution and non-significant dispersion ($P = .99$). Modelling showed a significant effect of the height of the high tide and the direction of the wind, along with the temporal and spatial scales on the abundance of beached birds (GLM $P < .05$, explained deviance = 46%). This metric was significantly higher as the height of the high tides increased (GLM $F_{1;160} = 2.4$, $P = .01$) along with westerly winds (GLM $F_{1;78} = 2.21$, $P = .02$). At the spatial scale, this metric was significantly higher in the upper forebeach (GLM $F_{1;160} = 7.92$, $P < .001$), but it was lower during the spring of 2021/22 at the temporal scale (GLM $F_{7;160} = -2.49$, $P = .01$) (Table 2).

The global model assessing the effect of the same set of predictor variables on overall beached bird richness showed a good fit,

with residuals following a normal distribution and non-significant dispersion ($P = .18$). The model indicated a significant effect of temporal and spatial scales on species richness (GLM $P < .05$, explained deviance = 42%). Specifically, species richness was lower during the spring of 2021/22 (GLM $F_{7;160} = -3.09$, $P = .001$) and higher in the upper forebeach (GLM $F_{1;160} = 7.00$, $P < .001$) (Table 2).

DISCUSSION

To our knowledge, this is the first study with a broad temporal scope to investigate beached birds in the northern sector of the San Matías Gulf, or elsewhere along the northern Argentine Patagonian coast. Our findings indicate that the Magellanic Penguin, Kelp Gull, and Neotropic Cormorant were the most abundant and frequently encountered species. This is likely related to the proximity of the study area to nesting grounds of these species: Novaro Island and the islets of Canal Escondido in the BSAPNA, and several islets within Lobos Islet National Park and at Punta Pozos. The population size of these species in these areas is likely important. For example, at a national level, the San Matías Gulf hosts < 1% of the Magellanic Penguin population (in terms of breeding pairs), 5.3% of the Kelp Gull population, and 29.3% of the Neotropic Cormorant population (see Frere et al., 2005; García Borboroglu et al., 2022 and references cited therein; González et al., 1998; Lisnizer et al., 2014; Pozzi et al., 2015). Other potential factors contributing to the dominance of just three species include their starvation-survival state and the presence of parasites (Ewbank et al., 2020; García et al., 2020, among others), which may increase vulnerability to anthropogenic disturbance, particularly the presence of feral and non-feral dogs on the coast (V. Pizá, personal observation, November 06, 2020).

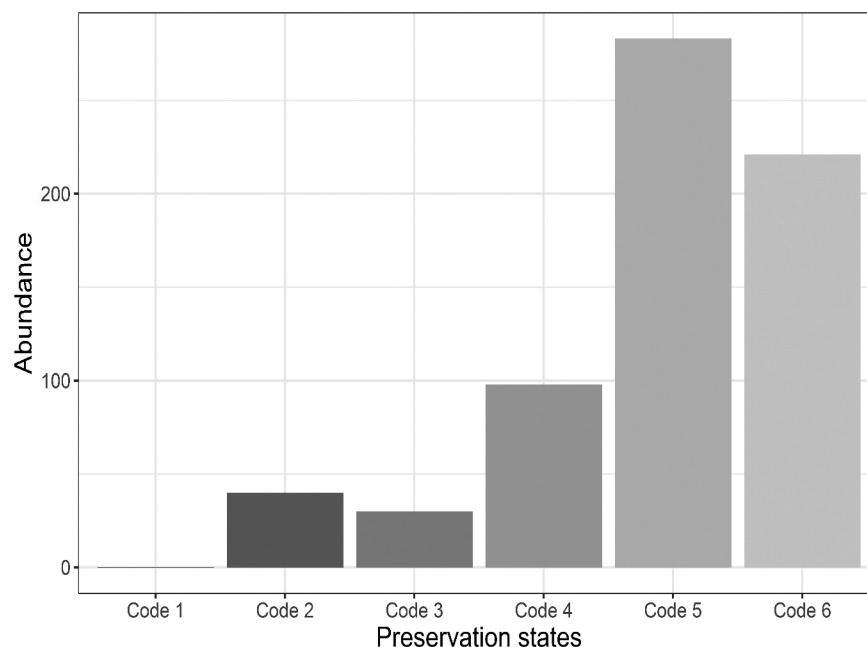


Fig. 2. Abundance (number of carcasses) of the specimens found beached according to their preservation status (scores). Scores were as follows: code 1, live animal with external wounds, signs of malnutrition and/or dehydration; code 2, recently dead birds, fresh odorless; code 3, mild decomposition, internal organs intact, little odor, no eyes; code 4, advanced decomposition, detachment of feathers, skin and/or scales, swollen body due to the accumulation of gases, alteration of the color, texture, and/or structure of the internal organs making their recognition difficult, strong odor; code 5, extremely advanced decomposition, dried body, presence of post-mortem wounds due to autolysis, parts of the skeleton visible, internal organs unrecognizable; code 6, carcasses composed of bones with or without remains of dry integument.

TABLE 2
Generalized linear model (GLM) results describing the relationship between abundance of beached birds and species richness, and the different explanatory variables ($n = 160$ transects)^a

Factors	Factor levels	Abundance				Richness			
		Estimate	SE ^b	Z value	P value	Estimate	SE ^b	Z value	P value
Levels of the beach									
	Upper forebeach	1.069	0.134	7.928	< .001*	0.817	0.116	7.007	< .001*
Season									
	Winter 2	-0.342	0.308	-1.110	.266	-0.488	0.254	-1.915	.055
	Autumn 1	0.118	0.273	0.431	.666	-0.154	0.212	-0.727	.467
	Autumn 2	-0.222	0.269	-0.826	.408	-0.332	0.210	-1.579	.114
	Spring 1	-0.177	0.303	-0.584	.559	-0.312	0.240	-1.299	.193
	Spring 2	-0.745	0.299	-2.493	.012*	-0.775	0.250	-3.098	.001*
	Summer 1	0.563	0.311	1.806	.07	0.335	0.233	1.434	.151
	Summer 2	0.298	0.300	0.992	.992	-0.152	0.244	-0.622	.534
Height of the high tide		0.523	0.218	2.401	.001*	0.325	0.179	1.821	.060
Tidal current intensity		-0.311	0.409	-0.761	.446	-0.214	0.330	-0.649	.516
Wave height		0.821	0.439	1.870	.061	0.228	0.341	0.670	.503
Direction of the wind									
	North	0.044	0.377	0.117	.906	-0.252	0.293	-0.861	.389
	West	0.757	0.417	1.815	.069	0.397	0.314	1.264	.206
	South	0.230	0.370	0.621	.534	-0.192	0.284	-0.677	.498

^a The levels taken as reference for the explanatory variables for beach level, season, and wind direction were 'lower level', 'winter1' and 'wind blowing from the east quadrant,' respectively. Seasons denoted by the number 1 are those included in the period 2020/21, whereas those depicted by the number 2 included the period 2021/22. *P* values in bold and with asterisk shows statistical significance.

^b SE = standard error

Bycatch in commercial fisheries could also play a role (González Zevallos & Yorio, 2006; Yorio & Caille, 1999, among others).

Other species among the beached assemblage belonged to the Order Procellariiformes, such as Manx Shearwater and Great Shearwater *Ardenia gravis*. These two shearwater species are largely long-distance migrants: the Manx Shearwater migrates transequatorially, with ranges spanning Canada, the USA, the Faroe Islands, Ireland, Norway, and Spain (Blake, 1977; Cramp & Simmons, 1977), while the Great Shearwater's range encompasses the Tristan da Cunha, Nightingale, Inaccessible, and Gough islands (Carboneras, 1992; Ryan, 2007). Beached shearwaters are particularly abundant along the southern coasts of Buenos Aires Province, further north in Argentina, compared to albatrosses and petrels (Tamini & Dellacasa, 2009). Our results align with this pattern. Pelagic seabirds likely become beached primarily due to starvation and fatigue associated with migration (Mariani et al., 2019; Taylor, 1999, among others).

Few studies have reported beached bird encounter rates for the western South Atlantic coast (but see Estévez et al., 2002). The average encounter rate observed in this study (1.35 birds/km) falls within the range of encounter rates reported in other South American studies. At a regional level, encounter rates may vary from 0.12 birds/km in southeastern Buenos Aires Province, northern Patagonia (Jorge, 2016), to 2.3 birds/km on the northern coast of Chile (Portflitt Toro et al., 2018), and 2.56 birds/km along the coast of Rio Grande do Sul, Brazil (Petry & Fonseca, 2002). The encounter rate for Magellanic Penguins (0.50 penguins/km) was

one order of magnitude lower when compared with that reported by Vanstreels et al. (2013; 1.32 penguins/km) for a period of seven years on the coast of Rio Grande do Sul. This could be related to the fact that the southern coast of Brazil receives a larger proportion of individuals during their post-reproductive migrations from Patagonia. To date, encounter rates of Kelp Gulls and Neotropic Cormorants have not been reported for the coast of the western South Atlantic, making regional comparisons impossible.

Nearly half of the carcasses we found were in a state of extremely advanced decomposition. This may be related to drift time in the sea, prolonged exposure to scavengers like the Kelp Gull, Chimango Caracara *Milvago chimango*, Crested Caracara *Caracara plancus*, Black Vulture *Coragyps atratus*, and Turkey Vulture *Cathartes aura* (V. Pizá, personal observation, March 26, 2021) both while in the water or after being deposited on the coast. Additional factors may include scavenging by terrestrial carnivores, feral dogs, and various arthropods (V. Pizá, personal observation, October 23, 2020).

A variety of factors, both natural and/or anthropogenic, may contribute to the wrecking of birds. These include (1) changes in the distribution and abundance of prey (Cairns, 1987; Diamond & Devlin, 2003), (2) factors intrinsic to the individual (such as sex, age class, and physical condition; Hindwood & McGill, 1955; Holmes, 1981; Mariani et al., 2019; Seco Pon & García, 2022), (3) maritime traffic (García Borboroglu et al., 2010; Perkins, 1983), (4) ingestion of and entanglement in marine debris (Jiménez et al., 2015; Kühn & van Franeker, 2020), and (5) injury due to interaction with fishing

operations (González Zevallos & Yorio, 2006; Oro et al., 2013). Climatic anomalies (Montevecchi & Myers, 1997; Trathan et al., 2015) and changes in environmental and oceanographic conditions (e.g., sea surface temperature and salinity, winds, wave height, currents, and tides) are considered the main natural stressors for facilitating the transfer of seabird carcasses to the coast (Haman et al., 2013; Wiese & Elmslie, 2006). Thus, studying beached birds requires approaches that consider both the oceanographic dynamics of the coastal area and the various causes of mortality, in order to gain a better understanding of their spatio-temporal patterns (Hart et al., 2006; Tavares et al., 2016).

Our analysis revealed a significant increase in beached bird abundance with higher high tides and specific wind directions. These findings align with previous studies (Bugoni et al., 2007; Tavares et al., 2020; Wiese & Jones 2001), although they are in contrast to others (Van Pelt & Piatt, 1995). These results reinforce the importance of conducting experiments to better understand the process of sinking, refloating, and the eventual arrival of dead bird carcasses on the coast. Additionally, particle transport simulation models, coupled with a baroclinic numerical model, could provide further insights (Saraceno et al., 2020; Tonini et al., 2013). Such studies should be accompanied by an assessment of the relationship between these processes and environmental data gathered *in situ*, including ocean circulation systems and local wind patterns (Bodkin & Jameson, 1991; Flint & Fowler, 1998; Hart et al., 2006).

This study confirmed that spatial scale influences both the abundance and richness of beached birds, with both metrics being higher in the upper forebeach. This spatial segregation may be a sampling artifact, as the beaches had a gentle slope toward the sea, while their backshore was protected from the strong winds by 10-m-high cliffs (Fucks et al., 2012; Isla et al., 2023). Thus, successive tidal cycles could accumulate carcasses at each high tide line, potentially affecting our records. Future studies are needed to assess the broader spatial extent of this pattern, taking tidal cycles (syzygy and quadrature tides) into account. Nevertheless, beached birds were consistently found during all seasons of the year, particularly those species that occur near the coast. This suggests that the wrecking of birds occurs regularly on the beaches of the Río Negro Province coast, indicating extensive avian use of the San Matías Gulf throughout the year, in accord with marine surveys (Curcio et al., 2017). Although the present study was conducted in a small coastal sector of the San Matías Gulf, it largely reflects the diversity of coastal and marine birds present in the region.

CONCLUSIONS

The data collected during beached bird surveys in northeastern San Matías Gulf were valuable for identifying trends in beached birds, as well as the environmental and oceanographic conditions that may explain some of the observed variation in the wrecking. To determine whether the data reported here are representative of the entire San Matías Gulf coastline or the broader maritime coasts of the Río Negro Province, it is crucial to expand the geographic scope of future beached bird surveys.

ACKNOWLEDGEMENTS

This work constitutes a partial fulfilment of the requirements for the degree of Bachelor's in Marine Science of VMP at the Universidad Nacional de Comahue – Escuela Superior de Ciencias

Marinas, Argentina. We thank the Secretaría de Ambiente y Cambio Climático of Río Negro Province for granting the permit (Res. N°16/2021, Exp. N°210378-SPDYS-2020) required to conduct this study. We also appreciate the valuable comments from the reviewers and editor, which significantly improved the manuscript.

REFERENCES

- Avery, G. (1984). Results of patrols for beached seabirds conducted in southern Africa in 1982. *Marine Ornithology*, 12(1), 29–43.
- Balech, E., & Ehrlich M. D. (2008). Esquema biogeográfico del Mar Argentino. *Revista de Investigación y Desarrollo Pesquero*, 19, 45–75.
- Batchelor, A. L. (1981). The August 1981 Seabird (*Pachyptila* and *Halobaena* spp.) wreck off Port Elizabeth, South Africa. *Marine Ornithology*, 9(2), 105–112. <http://doi.org/10.5038/2074-1235.9.2.71>
- Ben-Gigirey, B., Soliño, L., Bravo, I., Rodríguez, F., & Casero, M. V. (2021). Paralytic and amnesic shellfish toxins impacts on seabirds, analyses and management. *Toxins*, 13(7), 454. <https://doi.org/10.3390/toxins13070454>
- Blanco, D. E., & Canevari, P. (1995). *Situación actual de los chorlos y playeros migratorios de la zona costera Patagónica (Provincia de Río Negro, Chubut y Santa Cruz)*. Informe Técnico N° 3. Fundación Patagonia Natural. Plan de Manejo Integrado de la Zona Costera Patagónica. <https://patagonianatural.org.ar/wp-content/uploads/2021/04/244-IT03.pdf>
- Blake, E. R. (1977). *Manual of neotropical birds*. Vol. 1. University of Chicago Press.
- Bodkin, J. L., & Jameson, R. J. (1991). Patterns of seabird and marine mammal carcass deposition along the central California coast, 1980–1986. *Canadian Journal of Zoology*, 69(5), 1149–1155. <https://doi.org/10.1139/z91-163>
- Brusius, B. K., de Souza, R. B., Barbieri, E., & Forecast, W. (2020). Stranding of marine animals: Effects of environmental variables. In W. Leal Filho, A. M. Azul, A. M., L. Brandli, A. L. Salvia & T. Wall (Eds.), *Life below water*. Springer. https://doi.org/10.1007/978-3-319-71064-8_102-1
- Brusius, B. K., de Souza, R. B., Pereira de Freitas, R. A., & Barbieri, E. (2021). Effects of environmental variables on Magellanic penguin (*Spheniscus magellanicus*) strandings in southeastern Brazil. *Ocean and Coastal Management*, 210, 105704. <https://doi.org/10.1016/J.OCECOAMAN.2021.105704>
- Bugoni, L., Sander, M., & Costa, E. S. (2007). Effects of the first southern Atlantic hurricane on Atlantic petrels (*Pterodroma incerta*). *The Wilson Journal of Ornithology*, 119(4), 725–730. <https://doi.org/10.1676/06-141.1>
- Burger, J., & Gochfeld, M. (2004). Marine birds as sentinels of environmental pollution. *EcoHealth*, 1, 263–274. <https://doi.org/10.1007/s10393-004-0096-4>
- Byrd, B. L., Hohn, A. A., Lovewell, G. N., Altman, K. M., Barco, S. G., Friedlaender, A., Harms, C. A., McLellan, W. A., Moore, K. T., Rosel, P. E., & Thayer, V. G. (2014). Strandings as indicators of marine mammal biodiversity and human interactions off the coast of North Carolina. *Fishery Bulletin*, 112(1), 1–23. <https://doi.org/10.17615/50vs-hb24>
- Cabrera, A. I. (1976). *Regiones Fitogeográficas Argentinas*. Editorial Acme.
- Cairns, D. K. (1987). Seabirds as indicators of marine food supplies. *Biological Oceanography*, 5(4), 261–271. <https://doi.org/10.1080/01965581.1987.10749517>

- Camphuysen, C. J., & Heubeck, M. (2001). Marine oil pollution and beached bird surveys: the development of a sensitive monitoring instrument. *Environmental Pollution*, 112(3), 443–461. [https://doi.org/10.1016/s0269-7491\(00\)00138-x](https://doi.org/10.1016/s0269-7491(00)00138-x)
- Camphuysen, K., & Vollaard, B. (2016). Oil pollution in the Dutch sector of the North Sea. In A. Carpenter (Ed.), *Oil pollution in the North Sea*. Springer International Publishing. https://doi.org/10.1007/698_2015_438
- Carboneras, C., (1992). Family Procellariidae (Petrels and Shearwaters). In J. Del Hoyo, A. Elliot & J. Sargatal (Eds.), *Handbook of the birds of the world: Ostrich to ducks*. Lynx Edicions.
- Clarke, K. R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology*, 18, 117–143.
- Clarke, K. R., & Gorley, R. N. (2006). *PRIMER v6: User manual/tutorial (Plymouth Routines in Multivariate Ecological Research)*. PRIMER-E.
- Clarke, K. R., & Warwick, R. M. (2001). *Change in marine communities: An approach to statistical analysis and interpretation*. PRIMER-E.
- Cramp, S., & Simmons, K. E. L. (Eds.). (1977). *The birds of the western Palearctic*. Vol. 1. Oxford University Press.
- Crawley, M. J. (2007). *The R Book*. John Wiley and Sons. <https://doi.org/10.1007/s00362-008-0118-3>
- Croxall, J. P., Butchart, S. H. M., Lascelles, B., Stattersfield, A. J., Sullivan, B., Symes, A., & Taylor, P. (2012). Seabird conservation status, threats and priority actions: A global assessment. *Bird Conservation International*, 22(1), 1–34. <https://doi.org/10.1017/S0959270912000020>
- Curcio, N., Arias, M., Svendsen, G., Romero, A., & González, R. (2017). *Reporte de los Relevamientos Náuticos de Mamíferos y Aves Marinas a Bordo del Guardacostas GC69 “Río Paraná” – Resumen de los años 2014 al 2016*. Informe Técnico N°02-2017. San Antonio Universidad Nacional del Comahue, Escuela Superior de Ciencias Marinas.
- de la Barra, P., Iribarne, O., & Narvarte, M. (2019). Combining fishers’ perceptions, landings and an independent survey to evaluate trends in a swimming crab data-poor artisanal fishery. *Ocean & Coastal Management*, 173, 26–35. <https://doi.org/10.1016/J.OCECOAMAN.2019.02.008>
- Di Giacomo, A. S., & Coconier, E. (2007). Conservación de aves en Río Negro. In A. S. Di Giacomo, M. V. de Francesco, & E. G. Coconier (Eds.), *Áreas importantes para la conservación de las aves en Argentina – Sitios prioritarios para la conservación de la biodiversidad*. Aves Argentinas/Asociación Ornitológica del Plata.
- Diamond, A. W., & Devlin, C. M. (2003). Seabirds as indicators of changes in marine ecosystems: Ecological monitoring on Machias Seal Island. *Environmental Monitoring and Assessment*, 88, 153–181. <https://doi.org/10.1023/A:1025560805788>
- Diamond, A. W., McNair, D. B., Ellis, J. C., Rail, J.-F., Whidden, E. S., Kratter, A. W., Courchesne, S. J., Pokras, M. A., Wilhelm, S. I., Kress, S. W., Farnsworth, A., Iliff, M. J., Jennings, S. H., Brown, J. D., Ballard, J. R., Schweitzer, S. H., Okoniewski, J. C., Gallegos, J. B., & Stanton, J. D. (2020). Two unprecedented auk wrecks in the northwest Atlantic in winter 2012/13. *Marine Ornithology*, 48(2), 185–204. <http://doi.org/10.5038/2074-1235.48.2.1373>
- Estévez, J., Mameli, L., & Goodall, N. (2002). An expert system to help taxonomic classification in avian archaeology: A first attempt with bird species from Tierra del Fuego. *Acta Zoologica Cracoviensis*, 45, 383–391.
- Ewbank, A. C., Sacristán, C., Costa-Silva, S., Antonelli, M., Lorenzo, J. R., Nogueira, G. A., Ebert, M. B., Kolesnikovas, C. K. M., & Catão-Dias, J. L. (2020). Postmortem findings in Magellanic penguins (*Spheniscus magellanicus*) caught in a drift gillnet. *BMC Veterinary Research*, 16, 153. <https://doi.org/10.1186/s12917-020-02363-x>
- Flint, P., & Fowler, A. C. (1998). A drift experiment to assess the influence of wind on recovery of oiled seabirds on St. Paul Island, Alaska. *Marine Pollution Bulletin*, 36, 165–166.
- Frere, E., Quintana, F., & Gandini, P. (2005). Cormoranes de la costa Patagónica: Estado poblacional, ecología y conservación. *El Hornero*, 20(1), 35–52.
- Fucks, E. E., Schnack, E. J., & Charó, M. (2012). Aspectos geológicos y geomorfológicos del sector N del Golfo San Matías, Río Negro, Argentina. *Revista de la Sociedad Geológica de España*, 25, 95–105.
- Furness, R. W., & Camphuysen, K. (1997). Seabirds as monitors of the marine environment. *ICES Journal of Marine Science*, 54(4), 726–737. <https://doi.org/10.1006/jmsc.1997.0243>
- Garate, P. (2013). A mass mortality event of sooty shearwaters (*Puffinus griseus*) on the central coast of Peru. *Notornis*, 60, 258–261.
- García Borboroglu, P., Boersma, P. D., Ruoppolo, V., Pinho-da-Silva-Filho, R., Corrado-Adornes, A., Conte-Sena, D., Myiaji-Kolesnikovas, C., Dutra, G., Maracini, P., Carvalho-do-Nascimento, C., Ramos-Júnior, V., Barbosa, L., & Serra, S. (2010). Magellanic penguin mortality in 2008 along the SW Atlantic coast. *Marine Pollution Bulletin*, 60(10), 1652–1657. <https://doi.org/10.1016/J.MARPOLBUL.2010.07.006>
- García Borboroglu, P., Pozzi, L. M., Parma, A. M., Dell’Arciprete, P., & Yorio, P. (2022). Population distribution shifts of Magellanic Penguins in northern Patagonia, Argentina: Implications for conservation and management strategies. *Ocean & Coastal Management*, 226, 106259. <https://doi.org/10.1016/j.ocecoaman.2022.106259>
- García, G. O., Paterlini, C. A., Hernández, M. M., Behotas, R. T., Favero, M., & Seco Pon, J. P. (2020). Hematology and plasma chemistry values in beached Magellanic penguin (*Spheniscus magellanicus*) in northern Argentina during the nonbreeding season. *Journal of Zoo and Wildlife Medicine*, 50(4), 927–936. <https://doi.org/10.1638/2019-0012>
- Genchi, S. A., Carbone, M. E., Piccolo, M. C., & Perillo, M. E. (2010). Déficit hídrico en San Antonio Oeste, Argentina. *Revista de Climatología*, 10, 29–34.
- Geraci, J. R., & Lounsbury, V. J. (1993). *Marine mammals ashore: A field guide for strandings*. Texas A & M Sea Grant Publication.
- Geraci, J. R., & Lounsbury, V. J. (2005). *Marine mammals ashore: A field guide for strandings* (2nd ed.). Editorial National Aquarium in Baltimore.
- Giaccardi, M., & Reyes, L. (2012). *Plan de Manejo del Área Natural Protegida Bahía de San Antonio, Río Negro*. Programa de apoyo a la modernización productiva de la Provincia de Río Negro. Préstamo BIID 1463/1464 OC-AR. Gobierno de la Provincia de Río Negro.
- González, P. M. (2007). San Antonio Oeste. In A. S. Di Giacomo, M. F. de Francesco, & E. G. Coconier (Eds.), *Áreas importantes para la conservación de las aves en Argentina. Sitios prioritarios para la conservación de la biodiversidad*. Aves Argentinas/Asociación Ornitológica del Plata.
- González, P. M., Bertellotti, M., Giaccardi, M., Lini, R., Lizurume, M. E., & Yorio, P. (1998). Distribución reproductiva y abundancia de las Aves Marinas en Río Negro. In P. Yorio, E. Frere, P. Gandini, & G. Harris (Eds.), *Atlas de la distribución reproductiva de aves marinas en el litoral Patagónico Argentino*. Fundación Patagonia Natural & Wildlife Conservation Society.

- González, R., Narvarte, M., & Verona, C. (2010). *Principios, lineamientos generales y procedimientos para la elaboración, adopción, implementación, evaluación y revisión de los planes de manejo ecosistémico para la pesca marítima de captura en el Golfo San Matías*. ECOPEP (Iniciativa para un Ecosistema Pesquero SustenTable). Instituto de Biología Marina y Pesquera Almirante Storni. Universidad Nacional del Comahue.
- González Zevallos, D., & Yorío, P. (2006). Seabird use of discards and incidental captures at the Argentine hake trawl fishery in the Golfo San Jorge, Argentina. *Marine Ecology Progress Series*, 316, 175–183.
- Grémillet, D., Ponchon, A., Paleczny, M., Palomares, M. L. D., Karpouzi, V., & Pauly, D. (2018). Persisting worldwide seabird-fishery competition despite seabird community decline. *Current Biology*, 28(24), 4009–4013. <https://doi.org/10.1016/j.cub.2018.10.051>
- Guidi, C. (2019). *Análisis de la pesca recreacional, los usuarios y sus prácticas en tres pesqueros Norpatagónicos para aportar a su manejo* [Bachelor's thesis, Universidad Nacional de Río Negro, Sede Atlántica].
- Haman, K. H., Norton, T. M., Ronconi, R. A., Nemeth, N. M., Thomas, A. C., Courchesne, S. J., Segars, A., & Keel, M. K. (2013). Great shearwater (*Puffinus gravis*) mortality events along the eastern coast of the United States. *Journal of Wildlife Diseases*, 49(2), 235–245. <https://doi.org/10.7589/2012-04-119>
- Hamel, N. J., Burger, A. E., Charleton, K., Davidson, P., Lee, S., Bertram, D. F., & Parrish, J. K. (2009). Bycatch and beached birds: assessing mortality impacts in coastal net fisheries using marine bird strandings. *Marine Ornithology*, 37(1), 41–60. <http://doi.org/10.5038/2074-1235.37.1.810>
- Hart, K. M., Mooreside, P., & Crowder, L. B. (2006). Interpreting the spatio-temporal patterns of sea turtle strandings: going with the flow. *Biological Conservation*, 129(2), 283–290. <https://doi.org/10.1016/J.BIOCON.2005.10.047>
- Hartig, F. (2022). *DHARMA: Residual Diagnostics for Hierarchical (Multi-Level/Mixed) Regression Models* (R package Version 0.4.5) [Software]. <https://CRAN.R-project.org/package=DHARMA>
- Heubeck, M. (1995). Shetland beached bird surveys: national and European context. *Proceedings of the Royal Society of Edinburgh*, 103, 165–179. <https://doi.org/10.1017/S0269727000005996>
- Hindwood, K. A., & McGill, A. R. (1955). Sea-bird mortality in coastal New South Wales during July 1954. *Emu*, 55, 148–156.
- Holmes, O. (1981). Unequal sex ratios among seabirds found beach-washed. *Emu*, 81, 44–47. <https://doi.org/10.1071/MU9810044a>
- Instituto Nacional de Estadística y Censos. (2010). *Censo 2010*. Retrieved April 15, 2024, from <http://www.indec.gob.ar/indec/web/Nivel4-Tema-2-41-135>
- Isla, M. F., Moyano-Paz, D., Fitzgerald, D. M., Simontacchi, L., & Veiga, G. D. (2023). Contrasting beach-ridge systems in different types of coastal settings. *Earth Surface Processes and Landforms*, 48(1), 47–71. <https://doi.org/10.1002/esp.5429>
- Jiménez, S., Domingo, A., Brazeiro, A., Defeo, O., & Phillips, R. A. (2015). Marine debris ingestion by albatrosses in the southwest Atlantic Ocean. *Marine Pollution Bulletin*, 96(1–2), 149–154. <https://doi.org/10.1016/J.MARPOLBUL.2015.05.034>
- Jorge, D. M. (2016). *Ocurrencia y diversidad de vertebrados marinos varados en playas del partido de villa gesell, Provincia de Buenos Aires* [Bachelor's thesis, Universidad Nacional de Mar del Plata, Facultad de Ciencias Exactas y Naturales].
- Kokot, R., Salminci, J., Luna, F., & Tunstall, C. (2013). Retroceso costero y su relación con parámetros geotectónicos. Las Grutas, Río Negro. *Revista de Geología Aplicada a la Ingeniería y al Ambiente*, 30, 53–66.
- Korschenewski, P. (1975). Contribución al estudio del Pingüino Común o Magallánico (*Spheniscus magellanicus*). *El Hornero*, 11, 320–321.
- Kühn, S., & Van Franeker, J. A. (2020). Quantitative overview of marine debris ingested by marine megafauna. *Marine Pollution Bulletin*, 151, 110858. <https://doi.org/10.1016/J.MARPOLBUL.2019.110858>
- Larsen, J. L., Durinck, J., & Skov, H. (2007). Trends in chronic marine oil pollution in Danish waters assessed using 22 years of beached bird surveys. *Marine Pollution Bulletin*, 54(9), 1333–1340. <https://doi.org/10.1016/j.marpolbul.2007.06.002>
- Lisnizer, N., García-Borboroglu, P., & Yorío, P. (2014). Demographic and breeding performance of a new Kelp Gull (*Larus dominicanus*) colony in Patagonia, Argentina. *Ardeola*, 61(1), 3–14. <https://doi.org/10.13157/arla.61.1.2014.3>
- Llanos, F. A., Failla, M., García, G. J., Giobine, P. M., Carbajal, M., González, P. M., Barreto, D. P., Quillfeldt, P., & Masello, J. F. (2011). Birds from the endangered Monte, the Steppes and Coastal biomes of the province of Río Negro, northern Patagonia, Argentina. *Check List*, 7(6), 782–797. <https://doi.org/10.15560/11025>
- Mace, G. M., & Baillie, J. E. (2007). The 2010 biodiversity indicators: challenges for science and policy. *Conservation Biology*, 21(6), 1406–1413. <https://doi.org/10.1111/j.1523-1739.2007.00830.x>
- Malcolm, H. M., Osborn, D., Wright, J., Wienburg, C. L., & Sparks, T. H. (2003). Polychlorinated biphenyl (PCB) congener concentrations in seabirds found dead in mortality incidents around the British coast. *Archives of Environmental Contamination and Toxicology*, 45, 0136–0147. <https://doi.org/10.1007/s00244-001-0188-x>
- Mallory, M. L., Robinson, S. A., Hebert, C. E., & Forbes, M. R. (2010). Seabirds as indicators of aquatic ecosystem conditions: a case for gathering multiple proxies of seabird health. *Marine Pollution Bulletin*, 60(1), 7–12. <https://doi.org/10.1016/J.MARPOLBUL.2009.08.024>
- Mariani, D. B., Almeida, B. J. M., Febrônio, A. D. M., Vergara-Parente J. E., Souza, F. A. L., & Mendonça, F. S. (2019). Causas de mortalidade de aves marinhas encalhadas na costa do Nordeste do Brasil. *Pesquisa Veterinária Brasileira*, 39(7), 523–529. <https://doi.org/10.1590/1678-5150-PVB-5812>
- Martuscelli, P., Silva-e-Silva, R., & Olmos, F. (1997). A large Prion *Pachyptila* wreck in south-east Brazil. *Cotinga*, 8, 55–57.
- Mazio, C. A., & Vara, C. D. (1983). *Las mareas del golfo San Matías*. Servicio de Hidrografía Naval, Armada Argentina. Departamento de Oceanografía.
- Ministerio de Ambiente y Desarrollo SustenTable & Aves Argentinas. (2017). *Categorización de las aves de la Argentina (2015)*. Ministerio de Ambiente y Desarrollo SustenTable de la Nación y Aves Argentinas. <https://avesargentinas.org.ar/sites/default/files/Categorizacion-de-aves-de-la-Argentina.pdf>
- Montevecchi, W. A., & Myers, R. A. (1997). Centurial and decadal oceanographic influences on changes in northern gannet populations and diets in the north-west Atlantic: implications for climate change. *ICES Journal of Marine Science*, 54(4), 608–614. <https://doi.org/10.1006/JMSC.1997.0265>
- Morea, J. P. (2019). A framework for improving the management of protected areas from a social perspective: The case of Bahía de San Antonio Protected Natural Area, Argentina. *Land Use Policy*, 87, 104044. <https://doi.org/10.1016/J.LANDUSEPOL.2019.104044>

- Narosky, S., & Fiameni, M. A. (1986). Aves pelágicas en Costa Bonita, Buenos Aires, Argentina. *El Hornero*, 12, 281–285.
- Narosky, T., & Yzurieta, D. (2010). *Guía de identificación de aves de Argentina y Uruguay*. Asociación Ornitológica del Plata.
- Narvarte, M., González, R., & Filippo, P. (2007). Artisanal mollusk fisheries in San Matías Gulf (Patagonia, Argentina): An appraisal of the factors contributing to unsustainability. *Fisheries Research*, 87(1), 68–76. <https://doi.org/10.1016/j.fishres.2007.06.012>
- Narvarte, M., González, R., Medina, A., & Avaca, M. S. (2011). Artisanal dredges as efficient and rationale harvesting gears in a Patagonian mussel fishery. *Fisheries Research*, 111(1–2), 108e115. <https://doi.org/10.1016/j.fishres.2011.07.002>
- Naval Hydrography Service (n.d.). *Ciudad Autónoma de Buenos Aires, Argentina: Servicio de Hydrografía Naval*. Retrieved 15 June 2022, from <http://www.hidro.gov.ar/>
- Nevins, H. M., Benson, S. R., Phillips, E. M., de Marignac, J., DeVogelaere, A. P., Ames, J. A., & Harvey, J. T. (2011). *Coastal Ocean Mammal and Bird Education and Research Surveys (BeachCOMBERS), 1997-2007: Ten years of monitoring beached marine birds and mammals in the Monterey Bay National Marine Sanctuary*. Marine Sanctuaries Conservation Series ONMS-11-02. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Office of National Marine Sanctuaries, Silver Spring, MD. <https://mlml.sjsu.edu/beachcombers/wp-content/uploads/sites/35/2017/10/Beachcomber-1997-2007.pdf>
- Oro, D., Genovart, M., Tavecchia, G., Fowler, M. S., & Martínez-Abraín, A. (2013). Ecological and evolutionary implications of food subsidies from humans. *Ecology Letters*, 16, 1501–1514. <https://doi.org/10.1111/ele.12187>
- Ortiz-Alvarez, C., Guidino, C., Verhaegen, C., Alfaro-Shigueto, J., & Mangel, J. C. (2022). Lessons from 12 years of marine fauna stranding data in the south of Peru. *Environmental Monitoring and Assessment*, 194, 142. <https://doi.org/10.1007/s10661-022-09782-3>
- Paleczny, M., Hammill, E., Karpouzi, V., & Pauly, D. (2015). Population trend of the world's monitored seabirds, 1950–2010. *PLOS One*, 10, e0129342. <https://doi.org/10.1371/journal.pone.0129342>
- Perkins, J. S. (1983). Oiled Magellanic penguins in Golfo San Jose, Argentina. *Marine Pollution Bulletin*, 14(10), 383–387. [https://doi.org/10.1016/0025-326X\(83\)90603-3](https://doi.org/10.1016/0025-326X(83)90603-3)
- Petry, M. V., & Fonseca, V. D. S. (2002). Effects of human activities in the marine environment on seabirds along the coast of Rio Grande do Sul, Brazil. *Ornitologia Neotropical*, 13, 137–142.
- Phillips, E. M., Nevins, H. M., Hatch, S. A., Ramey, A. M., Miller, M. A., & Harvey, J. T. (2010). Seabird bycatch in Alaska demersal longline fishery trials: a demographic summary. *Marine Ornithology*, 38(2), 111–117. <http://doi.org/10.5038/2074-1235.38.2.895>
- Portflitt Toro, M., Miranda Urbina, D., & Luna Jorquera, G. (2018). Aves marinas varadas en la bahía de Coquimbo, norte de Chile: ¿Qué especies y cuántas mueren? *Revista de Biología Marina y Oceanografía*, 53, 185–193. <https://doi.org/10.22370/rbmo.2018.53.2.1292>
- Povedano, H. E. (2016). *Aves de la Provincia de Río Negro: Identificación, distribución, status*. Povedano H.E.
- Powlesland, R. G. (1986). Seabirds found dead on New Zealand beaches in 1984 and a review of fulmar recoveries since 1960. *Notornis*, 33(2), 171–184.
- Powlesland, R. G., & Imber, M. J. (1988). OSNZ Beach Patrol Scheme: Information and instructions. *Notornis*, 35(2), 143–153.
- Pozzi, L. M., Borboroglu García, P., Boersma, P. D., & Pascual, M. A. (2015). Population regulation in Magellanic Penguins: What determines changes in colony size? *PLOS One*, 10, e0119002. <https://doi.org/10.1371/journal.pone.0119002>
- R Core Team. (2021). *R* (version 4.3.1) [Computer software]. The R Foundation for Statistical Computing. <https://www.R-project.org/>
- Roletto, J., Mortenson, K., Harrauld, I., Hall, J., & Grella, L. (2003). Beached bird surveys and chronic oil pollution in Central California. *Marine Ornithology*, 31(1), 21–28. <http://doi.org/10.5038/2074-1235.31.1.553>
- Roman, L., Hardesty, B. D., Hindell, M. A., & Wilcox, C. (2019). A quantitative analysis linking seabird mortality and marine debris ingestion. *Scientific Reports*, 9, 1–7. <https://doi.org/10.1038/s41598-018-36585-9>
- Romero, M. A., Svendsen, G., Arias, M., & González, R. (2021). *Varamiento masivo de delfines comunes Delphinus delphis en la Bahía de San Antonio*. Informe Técnico N°05-2021. Universidad Nacional del Comahue, Escuela Superior de Ciencias Marinas. http://rdi.uncoma.edu.ar/bitstream/handle/uncoma/16578/Informe%20ESCIMAR%20-%20Varamiento%20Delfines%20comunes_final.pdf?sequence=1&isAllowed=y
- Ryan, P. G. (2007). *Field guide to the animals and plants of Tristan da Cunha and Gough Island*. Pisces Publications.
- Saad, J. F., Narvarte, M. A., Abrameto, M. A., & Alder, V. A. (2019). Drivers of nano-and microplanktonic community structure in a Patagonian tidal flat ecosystem. *Journal of Plankton Research*, 41(5), 621–639. <https://doi.org/10.1093/PLANKT/fbz045>
- Sagerup, K., Åsbakk, K., Polder, A., Skaåre, J. U., Gabrielsen, G. W., & Barret, R. T. (2014). Relationships between persistent organic pollutants and circulating immunoglobulin-Y in black-legged kittiwakes and Atlantic puffins. *Journal of Toxicology and Environmental Health*, 77(9–11), 481–494. <https://doi.org/10.1080/15287394.2014.886543>
- Saraceno, M., Tonini, M. H., Williams, G. N., Aubone, N., Olascoaga, M. J., Beron-Vera, M. J., Gonzalez, R., Soria, M., Saad, J. F., & Svendsen. (2020). On the complementary information provided by satellite images, lagrangian drifters, and a regional numerical model: A case study in the San Matias Gulf, Argentina. *Remote Sensing in Earth Systems Sciences*, 3, 123–135. <https://doi.org/10.1007/s41976-020-00039-6>
- Savigny, C. (2021). *Aves del Atlántico Sudoccidental y Antártida*. Ediciones LBN.
- Savigny, C., & Carbajal, M. (2015). El albatros corona blanca *Thalassarche steadi* (Falla, 1933) en la República Argentina. Primer registro confirmado por espécimen y notas sobre su distribución e identificación en el campo. *Nótulas Faunísticas (Segunda Serie)*, 180, 1–9.
- Scalise, A., Schnack, E., Fucks, E., Ahrendt, K., González, R. (2009). *Evaluación de alternativas para la conservación y manejo del frente costero en Las Grutas, Provincia de Río Negro*. Consejo Federal de Inversiones (CFI).
- Seco Pon, J. P., & García, G. O. (2022). Pingüino de Magallanes (*Spheniscus magellanicus*) en la costa norte de Argentina: ¿Evidencias de un sesgo sexual en aves juveniles varadas? *El Hornero*, 37(1), 65–77.
- Shumway, S. E., Allen, S. M., & Boersma, P. D. (2003). Marine birds and harmful algal blooms: Sporadic victims or under-reported events? *Harmful Algae*, 2(1), 1–17. [https://doi.org/10.1016/S1568-9883\(03\)00002-7](https://doi.org/10.1016/S1568-9883(03)00002-7)

- Simeone, A., Anguita, C., Daigre, M., Arce, P., Vega, R., Luna-Jorquera, G., Portflitt-Toro, M., Suazo, C. G., Miranda-Urbina, D., & Ulloa, M. (2021). Spatial and temporal patterns of beached seabirds along the Chilean coast: Linking mortalities with commercial fisheries. *Biological Conservation*, 256, 109026. <https://doi.org/10.1016/j.biocon.2021.109026>
- Stowe, T. J., & Underwood, L. A. (1984). Oil spillages affecting seabirds in the United Kingdom, 1966–1983. *Marine Pollution Bulletin*, 15(4), 147–152. [https://doi.org/10.1016/0025-326X\(84\)90236-4](https://doi.org/10.1016/0025-326X(84)90236-4)
- Tamini, L. L., & Dellacasa, R. F. (2009). *Project Seabird Argentina: Conservation through community involvement: final report*. British Petroleum Conservation Programme. https://www.conservationleadershipprogramme.org/wp-content/uploads/2024/12/020407F_Argentina_FinalReport_Project-Seabirds.pdf
- Tavares, D. C., de Moura, J. F., Merico, A., & Siciliano, S. (2020). Mortality of seabirds migrating across the tropical Atlantic in relation to oceanographic processes. *Animal Conservation*, 23(3), 307–319. <https://doi.org/10.1111/acv.12539>
- Tavares, D. C., de Moura, J. F., & Siciliano, S. (2016). Environmental predictors of seabird wrecks in a tropical coastal area. *PLOS One*, 11(12), e0168717. <https://doi.org/10.1371/journal.pone.0168717>
- Taylor, G. A. (1999). Seabirds found dead on New Zealand beaches in 1996. *Notornis*, 46, 434–445.
- Taylor, G. A. (2004). Beach patrol scheme: seabirds found dead on New Zealand beaches, 1997–1999. *Notornis*, 51, 176–190.
- Thomas, L. (1996). Monitoring long-term population change: why are there so many analysis methods? *Ecology*, 77(1), 49–58. <https://doi.org/10.2307/2265653>
- Thornthwaite, C. W. (1948). An approach toward a rational classification of climate. *Geographical Review*, 38(1), 55–94. <https://doi.org/10.2307/210739>
- Tonini, M. H., Palma, E. D., & Piola, A. R. (2013). A numerical study of gyres, thermal fronts and seasonal circulation in austral semi-enclosed gulfs. *Continental Shelf Research*, 65, 97–110. <https://doi.org/10.1016/j.csr.2013.06.011>
- Trathan, P. N., García Borboroglu, P., Boersma, D., Bost, C.-A., Crawford, R. J. M., Crossin, G. T., Cuthbert, R. J., Dann, P., Davis, L. S., De La Puente, S., Ellenberg, U., Lynch, H. J., Mattern, T., Pütz, K., Seddon, P. J., Trivelpiece, W., & Wieneck, B. (2015). Pollution, habitat loss, fishing, and climate change as critical threats to penguins. *Conservation Biology*, 29(1), 31–41. <https://doi.org/10.1111/COBI.12349>
- Van Pelt, T. I., & Piatt, J. F. (1995). Deposition and persistence of beachcast seabird carcasses. *Marine Pollution Bulletin*, 30(12), 794–802. [https://doi.org/10.1016/0025-326X\(95\)00072-U](https://doi.org/10.1016/0025-326X(95)00072-U)
- Vanstreels, R. E. T., Adornes, A. C., Canabarro, P. L., Ruoppolo, V., Da Silva-Filho, R. P., & Catão-Dias, J. L. (2013). Female-biased mortality of Magellanic Penguins (*Spheniscus magellanicus*) on the wintering grounds. *Emu - Austral Ornithology*, 113(2), 128–134. <https://doi.org/10.1071/MU12060>
- Vassallo, M. (2021). *Varamientos de tortugas marinas en el sector costero norte de la Provincia de Buenos Aires: Análisis de su contribución espacio-temporal e identificación de posibles causas* [Bachelor's thesis, Universidad Nacional de Mar del Plata].
- Veitch, C. R. (1978). Waders of the Manukau Harbour and Firth of Thames. *Notornis*, 25, 1–24.
- Wiese, F. K., & Jones, I. L. (2001). Experimental support for a new drift block design to assess seabird mortality from oil pollution. *The Auk*, 118(4), 1062–1068. [https://doi.org/10.1642/0004-8038\(2001\)118\[1062:ESFAND\]2.0.CO;2](https://doi.org/10.1642/0004-8038(2001)118[1062:ESFAND]2.0.CO;2)
- Wiese, F. K., & Elmslie, K. (2006). Underuse and misuse of data from beached bird surveys. *Marine Ornithology*, 34(2), 157–159. <http://doi.org/10.5038/2074-1235.34.2.706>
- Wilcoxon, F. (1945). Individual comparisons by ranking methods. *Biometrics Bulletin*, 1(6), 80–83. <https://doi.org/10.2307/3001968>
- Windguru. (n.d.) *Windguru Czech Republic*. Retrieved June 2022, from <http://www.windguru.cz/>
- Yorio, P., & Caille, G. (1999). Seabird interactions with coastal fisheries in northern Patagonia: Use of discards and incidental captures in nets. *Waterbirds*, 22(2), 207–216. <https://doi.org/10.2307/1522209>
- Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A., & Smith, G. M. (2009). *Mixed effects models and extensions in ecology with R*. Editorial Springer-Verlag.
- Žydelis, R., Small, C., & French, G. (2013). The incidental catch of seabirds in gillnet fisheries: a global review. *Biological Conservation*, 162, 76–88. <https://doi.org/10.1016/j.biocon.2013.04.002>