



Assessing the conservation value of nature reserves: terrestrial birds in Isla de los Estados (Staten Island) Provincial Reserve, Tierra del Fuego, Argentina

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Abstract: Existing networks of nature reserves contain a biased sample of biodiversity. In Patagonia Argentina, most nature reserves focus their protection objectives on a particular ecosystem, geoform or scenic value, and usually are located in inaccessible areas. However, unique species or assemblages could inhabit less protected ecosystems, areas or habitats, which could be threatened depending of management. In this study, we assessed the conservation value of different ecosystem types and areas (fjords) in Isla de los Estados Provincial Reserve (RPIE, Argentina), using birds as study case. We chose three fjords (east, central and west) and five ecosystems types (forests at low and high elevation, open-lands at low and high elevation, and sea coasts). Bird's assemblage richness, density, biomass, trophic level, migratory status, and use of strata per ecosystems and fjords were characterized in 75 points (3 fjords \times 5 ecosystems \times 5 replicates) and evaluated using ANOVA and multivariate methods. Also, Shannon (H') and Pielou (J) indices were estimated for fjords and ecosystems. Passerine was the most abundant group, being mainly residents, omnivorous and carnivorous-scavenger, and they were observed mainly flying or in the canopy. Assemblage structure and function varied with ecosystem types, with higher richness and biomass in coasts and open-lands than in other ecosystems, but with greater density in forests. Multivariate analyses showed conspicuous groups for forests and coast sampling units, with significant differences among all ecosystem types except between low and high forests. Also, east fjord significantly differed in density and biomass from the others, but west fjord also differed in structure, function and bird assemblage. We conclude that greater conservation value must be assigned to ecosystem types or areas inhabited by threatened species (as open-lands at high elevation) and highest richness and variety of use of strata (as sea coasts). However, bird assemblage patterns have particularities in less valuable ecosystems and areas, which also justify their importance for conservation, or at least, prescriptions of low impact uses and activities in the management planning. Nature reserves are opportunities to preserve endemic species, habitats or areas of special interest, as low latitude or unique isolated landscape communities, and ecosystems underrepresented in the network of local, regional or world protected areas.

Abbreviations: ANOVA–Analysis of Variance; CO–Sea Coasts (0 m.a.s.l.); DCA–Detrended Correspondence Analysis; FH–Forests at High elevations (100-400 m a.s.l.); FL–Forests at Low elevations (0-100 m a.s.l.); MRPP–Multi-Response Permutation Procedures; OH–Open-lands and at High elevations (100-400 m a.s.l.); OL–Open-lands at Low elevations (0-100 m a.s.l.); RPIE–“Isla de los Estados, Isla de Año Nuevo and adjacent islets” Provincial Reserve.

Nomenclature: We followed bird species nomenclature provided by South American Classification Committee (Remsen et al. 2019).

Introduction

Networks of nature reserves are usually considered the most valuable resource for conserving global biodiversity (Bruner et al. 2001). However, it has been documented that nature reserves are sited on places where warrant protection for some particular species (Prendergast et al. 1999), are largely limited to sites of higher elevation and less productive soils (Scott et al. 2001) or have been developed under a political impulse between countries (Martínez Pastur et al. 2016a). Consequently, existing networks of nature reserves contain a biased sample of biodiversity, typical of places in less suitable ecosystems for the exploitation of natural resources (Lindenmayer et al. 2000). For example, Marinaro

et al. (2015) show that protected areas in Northern Argentina Gran Chaco did not include higher biodiversity of birds and trees; Brooks et al. (2004) found that more than half of areas protected by the World Conservation Union (Groombridge 1993) were not explicitly designated for the conservation of biodiversity; and Baldi et al. (2017) suggest that environmental representativeness and biodiversity protection have a minor roll driving current protection at global and regional levels. This can result in protection areas that leave aside most of the regional biodiversity (Luque et al. 2011, Martínez-Harms and Gajardo 2008, Martínez Pastur et al. 2016a). Also, nature reserves include ecosystems with different conservation value (Cowling et al. 2003).

In Patagonia Argentina, most nature reserves focus their protection objectives on a particular ecosystem (e.g., forests in Tierra del Fuego National Park), geoform or scenic value (e.g., glaciers and landscapes in Perito Moreno National Park), and usually are located in inaccessible areas (e.g., near frontiers and far away from main routes) (APN 2007, Marinaro et al. 2012). However, several ecosystems (e.g., *Nothofagus antarctica* forests, scrub and native pastures) and areas (e.g., northern Tierra del Fuego archipelago) are relegated of nature reserves, despite they are inhabited by unique species or assemblages (Lencinas et al. 2005, 2008a, b). In addition, biodiversity can greatly vary in a region with the same type of vegetation, like in *Nothofagus* forests of the Tierra del Fuego archipelago, where significant changes exist along latitudinal and longitudinal gradients (Martínez Pastur et al. 2016b, Lencinas et al. 2017). Moreover, the multiple uses that are allowed in some nature reserves, as is usual in provincial reserves including productive and/or touristic and recreational exploitation, could threat biodiversity. Therefore, the evaluation of the conservation value of ecosystems and areas in a nature reserve could provide useful tools for their management.

In Tierra del Fuego, birds are the most abundant and diverse group of vertebrates in the terrestrial ecosystems (Lencinas et al. 2005) and they could be even more relevant because they occupy many ecological niches and key ecological roles (Díaz et al. 2005). Moreover, birds are considered good indicators of ecological changes, and have been associated with global changes, such as climate change (Regos et al. 2015), and local changes, as those motivated by land-use (Bastos et al. 2016). Many bird species are conspicuous, can be detected and identified with relatively simple methods, and are usually better known than other taxonomic groups. For these reasons, birds are considered ideal study objects to be able to understand the ecology of landscapes. In this sense, the individual responses of some species could be completely different. Therefore, beyond analyzing the changes in the richness and diversity of the communities, studies of birds as bioindicators should be designed taking into account the particularities of the species of the assemblage, evaluating, for example, their trophic functions in ecosystems (Rusch et al. 2005). Knowledge of species-specific habitat requirements will also help in the development of effective conservation strategies. Certain structures, such as *Nothofagus* forests, rarely constitute large continuous extensions and exhibit marked temporal and spatial variability in resources, therefore species that occur only in a specific habitat acquire greater ecological and conservation importance, while those that have less specialization loses relevance for management at landscape scale (Becerra Serial and Grigera 2005, Lencinas et al. 2005, 2009). Indeed, some landscapes could be important or irreplaceable to ensure the continuity of resident and migratory bird species (Pressey et al. 2007).

In global terms, it is known that mountains, remote islands, and low latitudes support less densities of species than elsewhere (Rosenzweig 1995), but at the same time, endemism in these sites is more pronounced (Stevens 1989, Gaston and Williams 1996). The Isla de los Estados (Staten

Island) Provincial Reserve in Tierra del Fuego, Argentina, emerges 24 km east of Isla Grande of Tierra del Fuego, as the extreme of the Andes Mountain. It has unique geographical location and scenic value, including exposed cliffs and protected fjords that include several different ecosystems as forests, shrublands, grasslands and peat bogs, which are refuge for several native and endangered species. The human occupation today is reduced to military settlements of 2-6 people, but increasing interest for international tourism motivated the opening of some areas to recreational activities for multiple uses (e.g., berth of boats or yachts, tourism, human settlement, hiking) (SADSyCC 2017). The objective of this study was to assess the conservation value of different areas and ecosystems in a nature reserve where multiple uses are allowed, using bird assemblages as study case. To achieve this objective, we answer the following questions: (i) how do the structure (species composition, richness, density, biomass, diversity indices, assemblage patterns) and function (trophic level, habitat use, migratory status) of birds change among areas along a longitudinal gradient and ecosystem types? and (ii) is it possible to identify areas and/or ecosystems with greater conservation value than others, using bird assemblages as indicators?

Methods

Study area

The study was conducted at the “Isla de los Estados, Isla de Año Nuevo and adjacent islets” Provincial Reserve (hereafter RPIE, Tierra del Fuego, Antártida e Islas del Atlántico Sur province) located in the extreme south of Argentina, (54°38' to 54°54' SL, 63°47' to 64°45' WL) (Fig. 1). It is the last manifestation of the Andes (0-823 m a.s.l.), with an area of 496 km² of extremely rugged and mountainous landscape (Ponce and Fernández 2014). Deeply indented and dissected fjords, bays and harbors make up the coastlines (Dudley and Crow 1983) that, together with the south and southwestern coasts of the Isla Grande of Tierra del Fuego, are the only sites with coastal marine forest formations in the country.

The RPIE has unique natural and historical features in this region of the world, harboring the famous Lighthouse at the End of the World that was immortalized in the story of Jules Verne. The island is located 24 km east of the “Parque Provincial Península Mitre” (Argentina) of the Isla Grande of Tierra del Fuego, separated from the strait of Le Maire, and is accompanied by other smaller islands and islets that make up the Nuevo Año archipelago, the largest being the Observatory Island located 6.5 km north of the Isla de los Estados. In addition, it is located approximately 120 km north from the “Reserva de la Biósfera Cabo de Hornos” (Chile). The management plan of this nature reserve (SADSyCC 2017) includes accessible areas by boats or yachts (fjords) for multiple uses (e.g., tourism, human settlement, hiking), with other inaccessible ones, restricted and excluded from all use and management, for the protection of specially endangered fauna (e.g., several penguin species, as *Eudyptes chrysocome*) or invaluable archaeological sites.

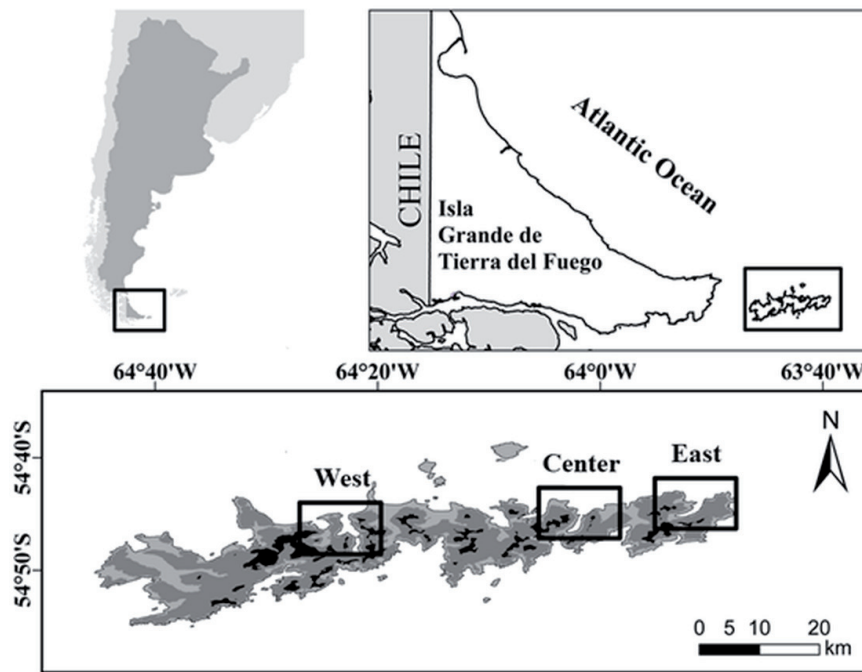


Figure 1. Isla de los Estados Provincial Reserve (RPIE), located in Tierra del Fuego, Antártida e Islas del Atlántico Sur province, Argentina. In light grey, lands with 0-100 m a.s.l.; in dark grey land with 100-400 m a.s.l.; and in black, more than 400 m a.s.l.

The RPIE is located phytogeographically in the Subantarctic province, within the Subantarctic Domain of the Antarctic Region (Cabrera 1976). The forests cover one-third of the surface of the island, with an area of 171 km². Fourteen percent of the whole surface is occupied by areas devoid of vegetation formed by rocky peaks, beaches and rocks in general. The lakes, which total 237, cover 3.4% of the total surface of the island. The rest is made up of different types of vegetative cover, predominantly peat bogs. The main soil type is Inceptisol (Cruzate and Panigatti 2007), while the ground along low and intermediate elevations is extremely wet, whereas the mountain peak ridges consist almost entirely of rocky promontories and mineral soil. The climate is predominantly sub-polar (Kottek et al. 2006), and strongly influenced by a persistent area of low pressure that develops near the Antarctic closeness (Ponce et al. 2017). Monthly mean temperature varied between 8.3°C (summer) and 3.3°C (winter), mean annual precipitation reaches to 1450 mm.year⁻¹ with strong frequently winds of 95-140 km.h⁻¹ (Dudley and Crow 1983). RPIE is currently very little intervened by humans and is probably one of the most remote and uninhabited places of Argentina.

Sampling design and data taking

Data collection was conducted during the breeding season for birds in 2014 (between late November to early December), covering the combination of two factors: (i) areas along a geographical longitudinal gradient, hereafter associated to different fjords (west, center and east) and (ii) ecosystem types (OL-Open-lands at Low elevations, OH-Open-lands at High elevations, FL-Forests at Low elevations, FH-Forests at High elevations and CO-sea COasts). Fjords were denominated: west, in the Hoppner Bay; center, in the Cook Bay; and east, in the San Juan de Salvamento Bay (Fig.

1). The three were chosen for their accessibility and representativeness in the mountain landscapes in the Island (Körner and Ohsawa 2005, Körner et al. 2011). Hoppner Bay is part of the Central Mountainous Massif, while Cook and San Juan de Salvamento Bays are in the Eastern Region of Fjords (SADSyCC 2017). The three fjords are in actually restricted use areas.

The studied ecosystem types were: open-lands (OL) and forests (FL) at low elevations (0-100 m.a.s.l.), with greater influence of sea closeness; open-lands (OH) and forests (FH) at high elevations (100-400 m.a.s.l.), close to the tree-line with greater exposure to extreme climate and mountain environments, and the sea coasts (CO) defined as the ecotone area between wooded and sea shore environments (0 m.a.s.l.). FL and FH are native forests without previous management, with *Nothofagus betuloides* as the dominant species. In FL trees are higher than in FH, with greater diameters and more bared soil due to greater closure of the canopy (Huertas Herrera et al. 2018). Open-lands included grasslands and peat-bogs, have a greater total cover of tussock and rhizomatous grasses and less woody debris than forests. Because of wind exposure and climatic limitations for fully vegetation development, OH presented more rocky outcrops and bared soil than OL, while total plant cover and richness are greater in OL than OH. Some species are dominant in both elevations, as *Massipospermum grandiflorum* and *Empetrum rubrum*, but others dominate in OL (as *Chilotrimum diffusum*, *Luzuriaga marginata* and *Pernettya pumila*, where *Caltha dioneifolia*, *Rubus geoides* and *Acaena pumila* are exclusive species), or only in OH (as *Astelia pumila* and *Bolax gummifera*, this last also an exclusive species of OH as *Festuca contracta* and *Azorella lycopodioides*) (Huertas Herrera et al. 2018).

A total of 75 bird observation points (sightings and tapping) were selected (3 fjords × 5 ecosystem types × 5 replicas) according to their homogeneity and accessibility. Each

observation point was established at least 150 m one from each other (300 m average distance between plots) depending on the size of the patch being considered into different ecosystem types and fjords. For this, plots were closer one to each other in ecosystem types that occupied small areas in fjords with very steep slopes, which produced a very difficult access to them (as FH in west fjord). Spatial closeness among samples can enclose some kind of autocorrelation due to mobility of individuals, but we avoid them performing sampling in closer plots not consecutively in time. We employed the method initially described in Lencinas et al. (2005) and successfully implemented in other studies (Lencinas et al. 2009, Martínez Pastur et al. 2015, Lencinas et al. 2018), which included a 10 minute observation period consisting of 2 minutes of habituation (time taken by birds to return to normal activity) and 8 minutes of counting. The 2 minute habituation is adequate since birds are not evasive (Deferrari et al. 2001). Sampling was carried out during the first 4 hours after dawn, due to the fact that it is the moments of greater social and feeding activity of the birds (Deferrari et al. 2001, Reyes-Arriagada et al. 2015). Sampling used a direct recognition method, identifying and counting the individuals of each seen species, without limit in the observation range (Lencinas et al. 2005). Distance to each individual was measured using a TruPulse laser rangefinder (Laser Technology, USA). Also, we registered each individual's use of strata (water, intertidal, forest soil, understory, shrub, tree trunk, tree canopy or flying).

Taxonomic classification follows South American Classification Committee (Remsen et al. 2019), while trophic level (herbivorous, insectivorous, carnivorous-scavenger, omnivorous, and marine omnivorous-carnivorous-piscivorous) was obtained from Humphrey et al. (1970), Lencinas et al. (2005) and Pizarro et al. (2012), and migratory status (resident or migrant) from Narosky and Yzurieta (1987) and Pizarro et al. (2012). Biomass of each bird species in grams was obtained from Dunning (1992).

Data analyses

The composition of the bird assemblages was analyzed considering shared and exclusive species of the different fjords and ecosystem types. The specific richness was calculated as the total number of species at each counting point, while density was estimated according to the methodological proposal of Lencinas et al. (2005). This methodology estimates density with the quantity of observed individuals in a circle area whose radius is the half of the maximum observation distance. This area is variable among ecosystem types, to account for decreasing in detectability with increasing distance (70.0 m in CO, 50.0 m in OL and OH, 16.0 m in FL and FH). Biomass was calculated as the sum of biomass of individuals obtained for the density, expressed as grams per hectare.

Regarding to the migratory status and the trophic level, we compared the proportional richness (% species) and abundance (% individuals, based on density values) for each category per fjord and ecosystem type; while for habitat use,

we only compared the proportional abundance. In addition, diversity was estimated by Shannon and Pielou indices for the five ecosystems and the three fjords. Shannon-Wiener diversity index was obtained as $H' = -\sum p_i \ln p_i$, where p_i is relative abundance of species i at each plot; while Pielou evenness index was obtained as $J = H'/H'_{\max}$, where $H'_{\max} = \ln(S)$, where S is richness from each patch (Pielou 1975).

In order to determine whether the variables of bird assemblages differed along a longitudinal gradient and ecosystem types we used a factorial analysis of variance (two-way ANOVA) after fitting to statistical assumptions (homogeneity, normality) were checked, in which fjords (west, center and east) and ecosystem types (OL, FL, OH, FH and CO) were the main factors. The response variables in all the analyses were: specific richness (number of species), density (individuals.ha⁻¹) and biomass (g.ha⁻¹). To meet the assumptions of ANOVA, variables were transformed prior to analysis using a transformation (richness) or transformation (density and biomass). Interaction terms (fjords × ecosystem types) were also analyzed, and when these were significant, differences among levels of one main factor for each level of the other main factor were evaluated by one-way ANOVA. Averages were tested for significant differences by Tukey test ($p < 0.05$). Statgraphics (Statistical Graphics Corp., USA) software was used for these analyses.

Three multivariate analyses were conducted with a matrix of density of bird species with more than 5% frequency (14 species × 58 samples) to analyze differences among fjords and ecosystem types:

(i) Detrended correspondence analysis (DCA) was performed to graphically evaluate the heterogeneity in species composition among ecosystem types. DCA was chosen due to provides simultaneously analyses of species and sampling units (Hill and Gauch 1980), allowing the examination of ecological interrelationships between them in a single-step analysis (Ludwig and Reynolds 1988). Also, the gradient length value was larger than four standard deviations (SD) indicating the convenience of the unimodal method (ter Braak and Šmilauer 2015). Analyses were done without down weight for rare species and with axis rescaling (Hill 1979).

(ii) Multi-Response Permutation Procedure (MRPP) was used to evaluate differences in assemblage groups among ecosystem types and fjords, based on Bray-Curtis distance and using T and p -value for evaluation (McCune and Grace 2002).

(iii) Indicator Species Analysis (Dufrêne and Legendre 1997) was employed to explore possible associations (in specificity and fidelity) of bird species with fjords and ecosystem types (e.g., Terraube et al. 2016). These analyses included a random reallocation procedure with 4999 permutations (Monte Carlo test) to evaluate the significance of the maximum indicator values (IndVal) provided ($p < 0.05$). Following Tejeda-Cruz et al. (2008), we considered as “indicator species” those species with IndVal > 50 and p values lower than 0.05.

All multivariate statistical analyses were performed using PC-ORD (McCune and Mefford 1999).

Table 1. Taxonomy, species code, migratory status, trophic level, individual biomass and occurrence frequency for the observed bird species in Isla de los Estados Provincial Reserve (RPIE).

Species	Code	Migratory status	Trophic level	Biomass (g)	Occurrence frequency (%)
<i>Chloephaga hybrida</i> *	CHHY	M	H	2888.5	10
<i>Lophonetta specularioides</i>	LOSP	R	M	1021.5	3
<i>Tachyeres patachonicus</i> *	TAPA	R	M	4200.0	2
<i>Cathartes aura</i>	CAAU	R	C	1430.0	10
<i>Vultur gryphus</i>	VUGR	R	C	11300.0	3
<i>Haematopus ater</i>	HAAT	R	M	724.0	3
<i>Larus dominicanus</i>	LADO	R	M	941.0	8
<i>Caracara plancus</i>	CAPL	R	C	1072.0	8
<i>Phalacrocorax magellanicus</i> *	PHAU	R	C	1187.0	10
<i>Zonotrichia capensis</i>	ZOCA	M	O	23.1	21
<i>Aphrastura spinicauda</i> *	APSP	R	I	12.3	39
<i>Cinclodes antarcticus</i> *	CIAN	R	C	63.2	2
<i>Cinclodes oustaleti</i>	CIOU	R	I	30.5	16
<i>Phrygilus patagonicus</i> *	PHPA	R	O	23.0	26
<i>Turdus falcklandii</i> *	TUFA	R	O	90.6	7
<i>Elaenia albiceps</i>	ELAL	M	O	16.5	8
<i>Muscisaxicola capistratus</i>	MUCA	M	I	14.7	8
<i>Muscisaxicola maclovianus</i>	MUMA	R	I	14.2	13
<i>Xolmis pyrope</i> *	XOPY	R	O	50.0	8
<i>Macronectes giganteus</i>	MAGI	R	M	3000.0	3
<i>Phalacrocorax brasilianus</i>	PHBR	R	M	1300.0	3
<i>Phalacrocorax magellanicus</i> *	PHMA	R	M	1485.0	3

Migratory status: R = resident and M = migrant. Dominant trophic level: C = carnivorous-scavenger, H = herbivorous, I = insectivorous, M = marine omnivorous-carnivorous-piscivorous, O = omnivorous. * = Endemic to the Patagonia region.

Results

A total of 22 species of 12 families of birds (Table 1) were identified across the whole study (none was exotic), being Passerine the group and Tyrannidae the family with most species. The most frequent species was the endemic *Aphrastura spinicauda* (39%), followed by *Phrygilus patagonicus* (26%) and *Zonotrichia capensis* (21%), while 64% of total richness (14 species) with less than 10% frequency in the whole sampling. Individual biomass varied between 11300.0 g to *Vultur gryphus* to 12.3 g to *A. spinicauda* (Table 1).

All fjords presented a very similar total richness (between 16 and 15 species), but center fjord presented the more diverse assemblage ($H' = 0.66$) following for east ($H' = 0.52$) and west fjords ($H' = 0.48$). Equitability (J) presented moderate values, varying between 0.55 (center and east) and 0.52 (west fjord). Concerning to the ecosystem types, CO presented the more diverse assemblage ($H' = 0.93$), followed by open-lands (H' varied between 0.41 and 0.75) and forests (H' between 0.21 and 0.47). The equitability of bird assemblages in the ecosystem types was very variable, with high values for CO and OL ($J = 0.73$ and 0.69 , respectively), medium

values for FL and OH ($J = 0.51$ for both), and low values for FH ($J = 0.21$).

Eight species were common among the three fjords, while three were shared between west and center fjord, four between center and east, and one between east and west (Fig. 2). The three fjords presented also exclusive species (one to three), with more exclusives in the west (Fig. 2a). In the case of the ecosystem types, six species were shared among ecosystems at low elevation (Fig. 2b), three between ecosystems at high elevation (Fig. 2c), three species were shared among forests (Fig. 2d) and eight among open-lands (Fig. 2e). *Chloephaga hybrida*, *Cinclodes antarcticus*, *Haematopus ater*, *Larus dominicanus*, *Lophonetta specularioides*, *Macronectes giganteus*, *Phalacrocorax brasilianus*, *P. magellanicus* and *Tachyeres patachonicus* appeared only in COs. While *Caracara plancus*, *V. gryphus* and *Xolmis pyrope* appeared only in open ecosystems. *A. spinicauda* and *P. patagonicus* were found in all ecosystem types (Fig. 2).

Resident species richness reached 82% considering the whole sampling. There was similar in west and east fjords (81% and 82% residents), but lower in center fjord (69% residents). Center fjord presented the greatest richness proportion

of migrant species (31%) compared to east and west fjords. Similar pattern was observed in proportional abundance per fjord and ecosystem types (Fig. 3), with most of 70% abundance corresponding to resident species. Concerning to the ecosystem type, migrant species represented less than 25% abundance in all of them (Fig. 3), with similar patterns for proportional richness except in FH, where there was the same richness proportion of migrant than resident species (data not shown).

Concerning to the trophic levels, the three fjords presented differences in the abundance proportion. In the center fjord, herbivore abundance was more abundant than in the other two fjords, while marine omnivore-carnivore-piscivore

abundance was greater in west fjord (Fig. 3), despite proportional richness was similar in these two trophic categories in all fjords. Contrary, both proportional abundance and richness of insectivorous birds were greater in west fjords, while center and east presented greater proportion in abundance and richness of omnivorous bird species. Concerning to the ecosystem types, insectivore (in FH) or insectivore and omnivore (in FL) were the most abundant in forests (Fig. 3), contrary to open lands where other trophic levels were dominant: omnivores in OL and carnivores and insectivores in OH. The CO was the unique environment where all the bird's trophic levels were observed, with herbivorous and marine omnivorous-carnivorous-piscivorous species. These two trophic levels

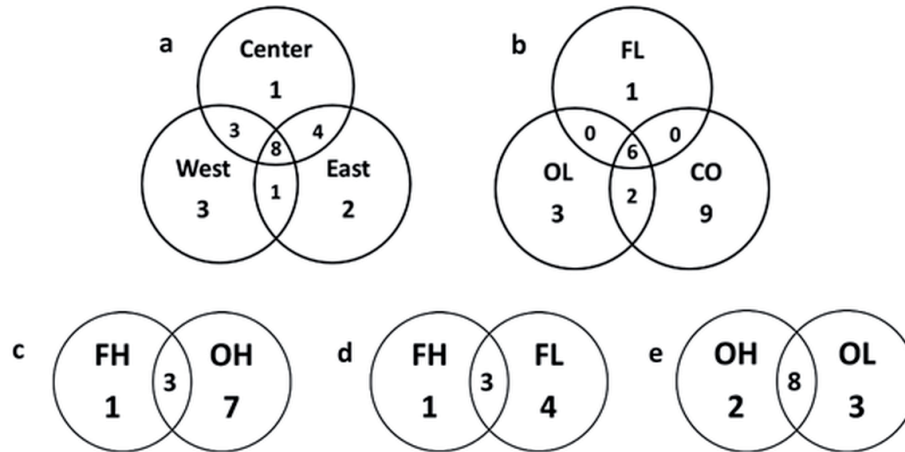


Figure 2. Species of birds shared and exclusive between (a) fjords, (b) low elevation ecosystem types, (c) high elevation ecosystem types, (d) forests and (e) open-lands. CO = coasts; OL = open-lands at low elevation; OH = open-lands at high elevation; FL = forests at low elevation; FH = forests at high elevation.

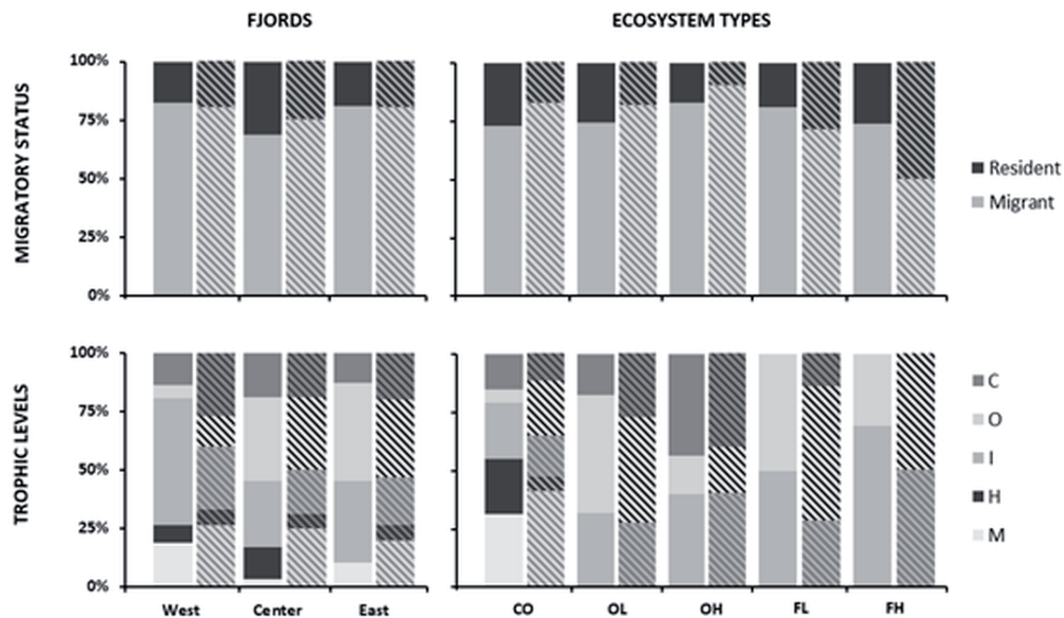


Figure 3. Proportion of bird abundance according to their (1) migratory status (migrant or resident) and (2) trophic levels (carnivorous-C, omnivorous-O, insectivorous-I, herbivorous-H, marine omnivorous-carnivorous-piscivorous-M) per fjords and ecosystem types. CO = coasts; OL = open-lands at low elevation; OH = open-lands at high elevation; FL = forests at low elevation; FH = forests at high elevation.

exceeded 50% of the abundance in said ecosystem and 40% of the richness.

Regarding to the use of strata, and considering the whole sampling, most birds were observed flying (36%) or in the tree canopy (28%). In west fjords, there were mainly observed flying, but in the east, they used more tree canopy. Meanwhile in the center fjord, there were almost the same abundance flying and in the canopy (Fig. 4). In the open-lands, individuals were more observed flying, in shrubs and canopy in OL, and shrubs and understory in OH, but in forests they used more tree canopy (Fig. 4). In the coast, birds were mostly found flying > intertidal > soil = water.

Two-way ANOVA did not detect significant differences among fjords for average bird richness ($F = 1.5$; $p = 0.23$; 2, 60 df), density ($F = 0.6$; $p = 0.54$; 2, 47 df) and biomass ($F = 0.3$; $p = 0.78$; 2, 47 df), but these were evident among ecosystem types (Table 2). Density was greater in forests than in other ecosystem types ($F = 32.0$; $p < 0.01$; 4, 47 df), while biomass was highest in CO than in forests and OL ($F = 4.2$; $p = 0.01$; 4, 47 df), presenting OH intermediate values (Table 2). We found a significant interaction between fjords and ecosystem types only for average richness ($F = 3.9$; $p < 0.01$; 8, 60 df), which occurred by significantly higher values in CO compared to OL, FL and FH in west fjord (OH with intermediate values), and significantly higher values in OL compared

to FL, FH and OH in center fjord (CO with intermediate values), while no differences were detected among ecosystem types for east fjords (Fig. 5).

In the graphical representation of the heterogeneity in species composition among ecosystem types by DCA, axis 1 (eigenvalue = 0.8447, length of gradient = 4.31) and axis 3 (eigenvalue = 0.2576, length of gradient = 3.42) were selected, being the total variance of 6.12 (Fig. 6). The graph showed clear grouping of forest sampling units (without split between high and low elevations) and coast ones, being open-land sampling units widely dispersed and slightly split among high and low elevations (Fig. 6a). When species were represented, those more typical from forests, like *A. spinicauda* (APSP), *Elaenia albiceps* (ELAL) and *P. patagonicus* (PHPA), and from coast, like *L. dominicanus* (LADO), *C. oustaleti* (CIOU) and *C. hybrida* (CHHY), were grouped (Fig. 6b). Evaluation of assemblage grouping by MRPP showed significant differences between fjords ($T = -3.68$, $p = 0.003$), generated by statistical differences in west fjord compared with center and east ($T < -3.08$, $p < 0.012$), mainly by the presence of LADO in low densities but exclusively in west fjord. Similarly, there were significant differences ($T = -14.67$, $p < 0.001$) in ecosystem types, presenting dissimilarities among all of them except between FL and FH ($T = -0.12$, $p = 0.354$), where APSP and PHPA were present with

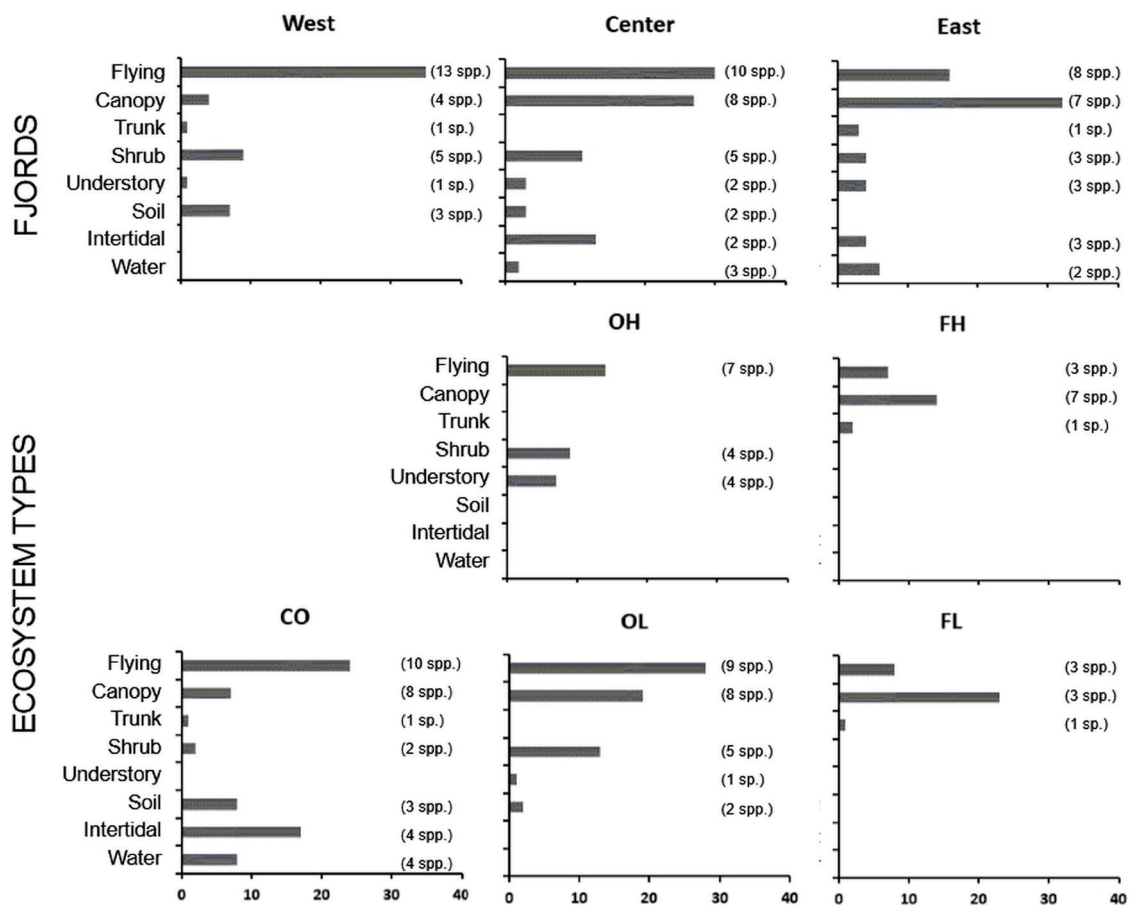


Figure 4. Abundance of birds by strata per fjord and ecosystem type, with corresponding species richness between parentheses. CO = coasts; OL = open-lands at low elevation; OH = open-lands at high elevation; FL = forests at low elevation; FH = forests at high elevation.

Table 2. Comparison between fjords and environments types for bird richness, density, biomass (Two-way ANOVA). Variables were transformed prior to analysis using \sqrt{x} for richness and \sqrt{x} for density and biomass. The averages are presented without transforming. $F(p)$ = F-test, with associated significance between parentheses; df = degrees of freedom (numerator, denominator). Different letters in a column indicate significant differences ($p < 0.05$) by Tukey. CO = coasts; OL = open-lands at low elevation; OH = open-lands at high elevation; FL = forests at low elevation; FH = forests at high elevation.

Factor	Treatment	Richness	Density (indiv.ha ⁻¹)	Biomass (g.ha ⁻¹)
A: Fjords	East	1.9	18.0	4.7
	Center	2.4	12.2	3.2
	West	1.8	6.4	1.6
	$F(p)$	1.5 (0.23)	0.6 (0.54)	0.3 (0.78)
	df	2, 60	2, 47	2, 47
B: Environment types	CO	3.1 b	5.5 a	7.9 b
	OH	1.5 a	2.5 a	2.2 a
	OL	2.5 ab	5.3 a	4.6 ab
	FL	1.3 a	25.8 b	0.7 a
	FH	1.8 a	21.7 b	0.3 a
	$F(p)$	5.6 (<0.01)	32.0 (<0.01)	4.2 (0.01)
	df	4, 60	4, 47	4, 47
Interaction AxB	$F(p)$	3.9 (<0.01)	2.12 (0.05)	1.1 (0.36)
	df	8, 60	8, 47	8, 47

Figure 5. Graphical representation of interactions for bird richness between fjords (west, center, east) and ecosystem types (CO, OL, FL, FH, OH), according to Table 2. Different capital letters indicate significant differences among ecosystem types for each fjord; while different lower case letters indicate significant differences among fjords for each level of ecosystem type, all by Tukey comparisons ($p < 0.05$). CO = coasts; OL = open-lands at low elevation; OH = open-lands at high elevation; FL = forests at low elevation; FH = forests at high elevation.

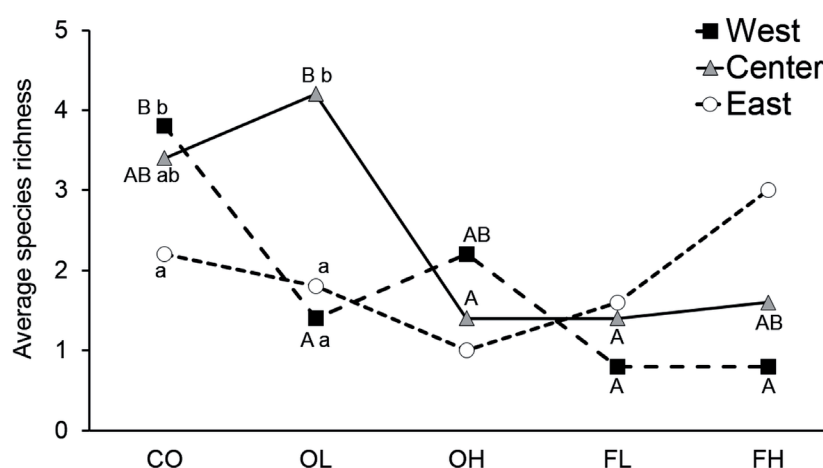
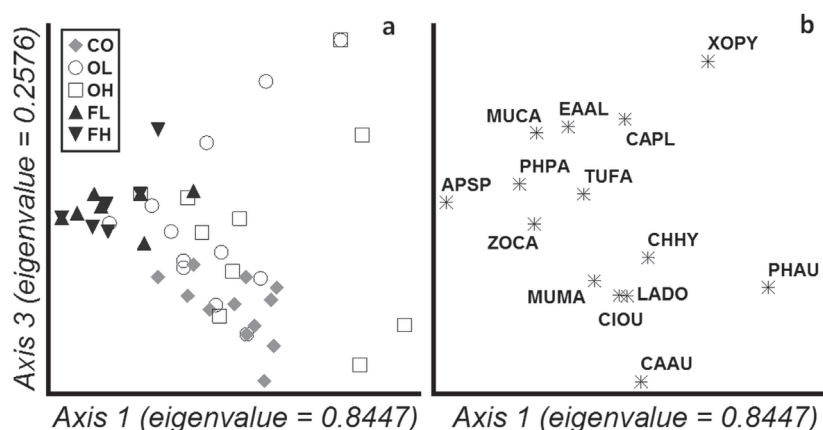


Figure 6. Heterogeneity in species composition among ecosystem types represented by DCA, with (a) sampling units and (b) bird species. See codes for the species in Table 1. CO = coasts; OL = open-lands at low elevation; OH = open-lands at high elevation; FL = forests at low elevation; FH = forests at high elevation.



more similar frequency and density, independently of the fjord. Regarding to the Indicator Species Analysis, two species were strictly emphasized as indicators: *C. hybrida* in CO and *Phalcoboenus australis* in OH, both with IndVal = 50.0 and $p < 0.001$. Beside this, no one species was highlighted as indicator of any particular fjord.

Discussion

Changes in structure and function of bird assemblages among fjords and ecosystem types

In general terms, center fjord presented the more rich and diverse assemblage of the observed bird species, despite significant differences were not found when comparing with east and center fjord. However, east fjord presented higher density and biomass, and center fjord presented the more different species assemblage, showing particularities for all areas along the longitudinal studied gradient. On the other hand, the coast represented the more diverse, equilibrated and highest biomass ecosystem type, but bird density was at least four times higher in forests than in coast or open ecosystems, with assemblage similarities among high and low elevations, highlighting the different function and structure of bird communities in each ecosystem type. Moreover, there is an interaction among ecosystem types and fjords that is mainly reflected in the average richness variable, which adopted the highest values for OL in center fjord and in CO for west fjord, while differences were not detected among ecosystem types in the east fjords. This emphasized the particularities of some ecosystem types in some fjords.

As is usual in remote islands, low latitudes and mountains (Rosenzweig 1995), RPIE presented a low bird richness (22 species in our study) in contrast to other sites of the world like upper Amazonia with around 500 species of birds (Orme et al. 2005, Haffer 1990). However, in our study almost half of the species are endemic to the Patagonia region, as was observed for other authors at low latitudes (Stevens 1989). In regional terms, the richness found in this work was low compared with those found in Cabo de Hornos archipelago (Chile), probably due to differences in size of the sampled area (Venegas Canelo 1991), or because other works include also sea bird species sampling (43 observed species in sea and terrestrial ecosystems according to Chebez and Bertonatti 1994). However, we cannot find other works specifically developed to characterize terrestrial bird communities in RPIE. Beside this, our results were comparable to those performed in terrestrial ecosystems of the Isla Grande of Tierra del Fuego (Deferrari et al. 2001, Lencinas et al. 2005, 2009, 2018, Martínez Pastur et al. 2015) and other zones of southern Chile (Venegas Canelo 1976, Diaz et al. 2005). On the other hand, the reported biomass and density here was lower to those found in of other studies in the region (Pizarro et al. 2012, Martínez Pastur et al. 2015).

Two of the most frequent species of the study, *A. spinicauda* and *P. patagonicus*, were very abundant in forests of southern South America (Rozzi et al. 1996). *A. spinicauda* is

resident endemic species of this region and the most abundant in many studies of temperate forests of Chile and Argentina (Deferrari et al. 2001, Lencinas et al. 2005, Martínez Pastur et al. 2015, Jiménez 2000, Ippi et al. 2009). *P. patagonicus* ranges from sea level to 1800 m.a.s.l. and is most often seen in forests (McGehee and Eitner 2007), forest borders or shrubby cleared areas in southern Chile and Argentina (Ridgely and Tudor 1989, Vuilleumier 1991). On the other hand, *Z. capensis* has one of the largest distributions of any Neotropical passerine, occurring from southern Mexico to almost the entire continent of South America (Chapman 1940). Usually it is common to abundant, found in open or semiopen areas and numerous in mountains (Ridgely and Tudor 1989). This species appears dominating harvested forests (Deferrari et al. 2001), associated with specific understory conditions, such shrubs (Martínez Pastur et al. 2015).

East fjord presented more density and biomass of birds than other fjords (Table 2), which could be related to their accessibility for coast and terrestrial bird species, as well as to the quality of the landscape matrix (Rayner et al. 2014) in which each fjord is embedded, or the availability of habitats that could be favorable for bird activities (Lencinas et al. 2018). In this fjord, the great use of canopy coincided with the largest density of trees in the forests, which presented greater height and diameter than in the other fjords, as was observed by Huertas Herrera et al. (2018). In center fjord, the greatest abundance of herbivorous birds coincided with the largest cover of shrubs and herbs noted by Huertas Herrera et al. (2018), but it seems did not depend of the grass cover, which was almost the same in the three fjords. In west fjord, no bird was observed on the coast or water and more than half of the individuals were observed flying. This is probably the reason because this greatly differed from the other fjords in MRPP analysis. This could be related to a greater difficulty of access to the coast by water than in the other fjords, due to their narrowness, exposure to strong winds, marine currents, tides and rock presence below the water. In contrast to what was expected, the observed bird density does not seem to diminish but to increase due to the tourist presence, since tourism is only allowed in Cook (center fjord) and San Juan de Salvamento (east fjord) Bays, where the bird density was high. However, it is necessary to carry out specific studies to corroborate a synergy between tourism and birds.

At the ecosystem type level, more species were found in the coasts, where marine and terrestrial environment join, and in open-lands than in forests, mainly at low elevations. The latter coincides with that observed by Vuilleumier (1998) in Chilean Fuego-Patagonia, where steppes had more species than forests, contrary to the theory that proposes structurally more complex habitats contain more bird species (MacArthur and MacArthur 1961). Concerning to coasts, as Venegas Canelo (1991) explains for the Cabo de Hornos archipelago, the greater specific richness and diversity that was found in the coast would result from a greater nutritional supply in the marine environment compared to an increasingly impoverished terrestrial environments as it advances in southern latitude. This greater nutritional supply should be also greater in

west fjord than in the other fjords, which could explain the interaction found in the ANOVA.

In relation to the trophic guilds, as Pizarro et al. (2012) found on Navarino Island, the coasts presented more carnivores and herbivores, while in terrestrial ecosystems, omnivores and insectivores were dominant, with the exception for OH that also presents carnivores. Coinciding with those found by Lencinas et al. (2005) in central zone of the Isla Grande of Tierra del Fuego, insectivores was the most important group on terrestrial bird assemblages, and carnivores were scarce in forests. Regarding to the migratory status, our results are coincident with those from other works in the region (Pizarro et al. 2012, Martínez Pastur et al. 2015), who also found that most of the species in the bird assemblage were residents. However, the high proportion of migratory birds in some ecosystems of this nature reserve (as FH and CO) could indicate more favorable conditions for migratory vs. resident competition (likely for resources such as food, refuge, and nesting sites).

Finally, spatial closeness among samples could imply some kind of autocorrelation, which can be considered a pseudoreplication due to the small sample size (Hurlbert 1984). In this work, only few plots were closer enough to have a higher probability of lack of independency between them, particularly in FH of the west fjord. However, spatial autocorrelation was diminished splitting samplings in time in closer plots (performing samplings in not consecutive time periods). Moreover, since spatial correlation could influence over richness, abundance or assemblage composition (e.g., Koenig 2001), it would be recommendable to consider it in more specific studies, since it should help to elucidate influences across space (Fleishman and MacNally 2006) and offers many different insights of ecological patterns and processes (Tobin 2004).

Identification of areas and ecosystems with greater conservation value

RPIE allowed us to test ecological responses in short distances, as suggested by Körner (2007), identifying coast ecosystems as those with more richness of species and trophic levels, center fjords as those with more used strata, and only two species exclusively associated to one ecosystem type: *C. hybrida* with coasts and *P. australis* with open high ecosystems. Therefore, coasts and open-lands presented more exclusive species (species with higher habitat requirements), indicator species, and greater diversity in terms of habitat use and trophic level than forests, highlighting the importance of these habitats for birds in terms of conservation value. *P. australis* has a very limited distribution, and is classified as Near Threatened (BirdLife International 2016) with an extant population of < 2,500 mature individuals (Balza et al. 2017). Furthermore, center fjord could be considered with greater conservation value than west and east fjords because of the higher diversity indices.

However, when we explore the assemblage pattern, each ecosystem type has particular characteristics (e.g., migra-

tory/resident status proportion, trophic level or use of strata) that could not be ignored to assign a conservation value. Similarly, Marinero et al. (2015) stated that territorial conservation strategies must include a combination of land tenures and habitat qualities derived from different land use practices and geographic locations, since they had particular indicator species. This supports the idea that studies of birds as bioindicators should be designed taking into account the particularities of the species in the assemblage, and not only evaluating their structure (Rusch et al. 2005). RPIE is an opportunity to preserve endemic species of the Patagonia region, areas of special interest or with unique characteristics considering their isolated surrounding landscape and geographic location in an extremely low latitude in a predominant oceanic hemisphere, other ecosystems also than forests (this last already present in other nature reserves in Argentinean and Chilean Patagonia, as Tierra del Fuego National Park), and very particular areas (or fjords) with unique combination of shape, elevation and topography that result in different resource offers and assemblages of biodiversity. Our analysis allowed to identifying ecosystems with greater conservation value, like coasts and open-lands, but since the bird species diversity is usually highly sensitive to the quality of the landscape context (Rayner et al. 2014), we cannot recommend to focus the conservation efforts and management planning only in these ecosystems, but in the whole Isla de los Estados Provincial Reserve. Nevertheless, more research is needed about the impact of the different uses and activities included in management planning to specifically ensure these are not threatening the RPIE biodiversity.

Conclusion

Greater conservation value must be assigned to those ecosystem types or areas inhabited by near threatened bird species (as open-lands at high elevation), and with highest richness and variety of use of strata (as sea coasts). However, bird assemblage patterns have particularities related to migratory status or trophic level in less valuable ecosystems and areas, which also justify their importance for conservation, or at least, prescriptions of low impact use and activities in the management planning. Nature reserves are opportunities to preserve endemic species, habitats or areas of special interest, as low latitude or unique isolated landscape communities, and ecosystems underrepresented in the network of local, regional or world protected areas.

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