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# Multi-taxon biodiversity assessment of Southern Patagonia: Supporting conservation strategies at different landscapes

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ABSTRACT

In the last years, different spatial analyses were developed to support multi-taxon biodiversity conservation strategies. In fact, the use of species distribution models as input allowed to create spatial decision-support maps. Of special interest are maps of potential biodiversity (MPB), which define distribution and ecological requirements of relevant species and maps of priority conservation areas (MPCA), which define priority areas considering endemism and richness. The objective of this paper was to assess multi-taxon biodiversity based on two different spatial analyses and to test their efficiency to support conservation decision at Patagonia. We computed 119 potential habitat suitability maps (one deer, birds, lizards, darkling-beetles, plants) with ENFA (Environmental Niche Factor Analysis) and 15 environmental variables, using Biomapper software. ENFA calculate two ecological indexes (marginality and specialization) which describe the narrowness of species niches and how extreme are the optimum environmental conditions related to the whole study area. These maps were combined obtaining a MPB and MPCA using Zonation software. Multivariate analyses were performed to compare methodologies, analysing environmental variables, ecological areas, forest types and protected areas. Multivariate and ecological indexes showed that deer, lizards and darkling-beetles presented a narrow range, while birds and plants presented a large range of marginality and specialization mainly related to vegetation and climate. At provincial level, highest potential biodiversity and conservation priority values were related to shrublands and humid steppes. However, MPCA showed higher values related to forests and alpine vegetation due to endemism, while MPB showed differences among forest types. These analyses showed that the most valuable areas were not represented in the protected areas, however, many higher conservation priority values were found inside the protected compared with unprotected areas. Different spatial decision-support maps presented similar outputs at provincial scale, but differed in the forest landscape matrix. Both methodologies can be used to plan conservation strategies depending on the specific objectives (e.g. highlighting richness or endemism).

## 1. Introduction

Global natural protected areas represent less than 10% of Earth's surface (Watson et al., 2014; UNEP-WCMC, IUCN and NGS, 2018) being the most traditional tool for nature conservation strategies (Miu et al., 2020). The concept of protected areas has been developed and refined along the years. At the beginning, these areas were mainly set up to protect iconic landscapes and rare, threatened or endemic taxa, and usually located in areas with few economic potentials uses (Watson et al., 2014; Venter et al., 2018). During the last years, ecosystem functions, services, and their importance to maintain human well-being, have been incorporated into the decision-making to protect biodiversity as a core of the conservation strategies (Bottrill and Pressey, 2012; Adams et al., 2019; Ramel et al., 2020). Biodiversity is one of the major

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drivers of ecosystem functions in the ecosystem services supply (MAES, 2013). However, land-sparing strategies (e.g. through protected areas) have been considered ineffective to conserve all the biodiversity under climate change and land use change scenarios (Watson et al., 2014; Reside et al., 2018). Therefore, other strategies (e.g. land-sharing, moveable protected areas) are necessary to improve the effectiveness of biodiversity conservation in unmanaged and managed landscapes (Carvalho et al., 2011; Gillson et al., 2013; Crespin and Simonetti, 2019).

In this context, statistical modelling is helpful to understand how biodiversity changes in the landscape and to achieve an effective design of new natural protected areas (Tulloch et al., 2016). Species distribution models (e.g. ENFA, GLM) are used in different software (e.g. Biomapper, MaxEnt, R) to define distribution maps (Guisan and Zimmermann, 2000; Hirzel et al., 2004a) by including assessments of biodiversity values and hotspot areas (Sofaer et al., 2019). Moreover, those analyses provide models of potential habitability (Hirzel et al., 2001, 2004b) by defining potential distribution and ecological requirements of species based on niche ecology concept (Hirzel and Le Lay, 2008). ENFA (Environmental Niche Factor Analysis) links environmental characteristics (e.g. climatic, topographic, landscape) and occurrence of species in a particular area (Guisan and Zimmermann, 2000; Hirzel et al., 2002) to determine areas of potential habitat suitability. In Biomapper software, ENFA uses geographical information data (e.g. remote sensing, climate and land use data) and only presence data (Hirzel et al., 2006), being a powerful tool for remote areas (e.g. Patagonia) with low available species data (Rosas et al., 2017). In addition, this software use a cross-validation analyses when database is only presence and compare the model results with a random modelling considering Boyce index (B), continuous Boyce index (Bcont), proportion of validation points (P), absolute validation index (AVI) and contrast validation index (CVI) (Boyce et al., 2002; Hirzel and Arlettaz, 2003; Hirzel et al., 2004b, 2006). However, it is necessary to identify areas for multi-taxon biodiversity to improve conservation actions, and not only for one charismatic or threatened species (Thomassen et al., 2011; Tulloch et al., 2016; Sofaer et al., 2019). In this sense, most of the studies combine multiple potential habitat suitability (e.g. average of values) to create maps of potential biodiversity for several species considering only one taxon (e.g. herb plants) (Martínez Pastur et al., 2016; Rosas et al., 2018, 2019a; 2019b; 2021a), while other authors used spatial decision-support software (e.g. Marxan, Zonation) (Moilanen et al., 2011; Daigle et al., 2020). Zonation software creates maps of priority conservation areas considering endemism or richness species (Moilanen et al., 2005; Moilanen, 2007). This methodology ranks conservation priority areas by iterative removal of the least important pixels (Di Minin et al., 2014). However, it is possible to improve those methodologies and combine different taxonomic groups (e.g. mammals, birds) to assess multi-taxon biodiversity. Both maps can be used as spatial decision-support maps: (i) to identify potential hotspot areas of biodiversity (Thomassen et al., 2011; Rosas et al., 2019b), (ii) to analyses the natural networking representativeness (Rosas et al., 2018, 2019a; Sofaer et al., 2019), (iii) to prioritize areas for biodiversity protection considering landscape connectivity (Daigle et al., 2020) at different spatial scales (Lehtomäki et al., 2009; Khosravi et al., 2019; Miu et al., 2020), their relationship with ecosystem services supply (Ramel et al., 2020; Rosas et al., 2021a) and under different climate change scenarios (Carvalho et al., 2011).

Santa Cruz province, located at South Patagonia, presents well-conserved wilderness areas dominated by steppe grasslands and native forests, which growing in a narrow strip along the Andean mountains. Protected areas are mostly located in these mountain areas (e.g. Los Glaciares National Park) in close contact with provincial and private protected areas or with protected areas in Chile (Fasioli and Díaz, 2011). In Argentina, the creation of national parks started in 1937, and most of them were located in isolated mountain landscapes (Watson et al., 2014), focused on unique values (e.g. ice fields), strategic geopolitical areas (e.g. international borders), uninhabited areas without significant economic interest, and with the preference for conserving *Nothofagus* forests over non-forested environments (Martín and Chehébar, 2001). *Nothofagus* forests (4125 km2) present three different forest types in Santa Cruz. *N. pumilio* forests (2246 km2) present a continuous distribution from north and central areas of the province, while mixed evergreen forests (180 km2) prevail close to humid site and low elevation near to big lakes, ice fields and *N. pumilio* forests in central areas (Peri et al., 2019). Mixed evergreen forests present a dominant cover of *N. betuloides* forests, associated with others species as *Drimys winteri, Embothrium coccineum, Maytenus boaria* and *N. pumilio*. *N. antarctica* forests (1699 km2) occur in small areas near to *N. pumilio* forests and dominate in the southern areas and occupy diverse environments such us mild slopes, hills, glacial moraines, plains and valleys with flooded soils (Peri and Ormaechea, 2013). *Nothofagus* forests present 48% of the area inside of protected networking areas (54% in national parks and 46% in provincial reserves). However, forest types are not equally included in protected areas networking, where 84% of the protected forests belongs to *N. pumilio* and 9% to mixed evergreen forests with different recreational activities (e.g. tourism) (Peri et al., 2019). In addition, only 7% of protected forests belongs to *N. antarctica* forests, where important economic activities prevail (e.g. livestock) (Peri et al., 2016a). In this context, it is necessary to develop a more representative and effective protection system to preserve high multi-taxon biodiversity values and ecosystem services including other vegetation types (e.g. steppes or wetlands) (Martín and Chehébar, 2001). Some studies using different methodologies (e.g. richness and endemism) identified areas to preserve particular species groups (e.g. lizards, darkling-beetles) in the Patagonian steppes (Carrara and Flores, 2013; Breitman et al., 2014), however, only one study considered different taxa (e.g. mammals and birds) at landscape level (Chehébar et al., 2013). In this context, the objective of this paper was to assess multi-taxon biodiversity based on two different spatial analyses and to test their efficient to support conservation decision in Santa Cruz province (Patagonia, Argentina).

## 2. Materials and methods

### 2.1. Study area

The study area is the whole Santa Cruz province (Argentina) (46°00′ to 52°30′ S, 66°00′ to 73°00′ W) (Fig. 1a). Protected areas represented 7% of the province (Fasioli and Díaz, 2011), where national parks mainly preserve specific landscapes in the west (e.g. Perito Moreno National Park), and provincial reserves mainly protect breeding areas of migrant birds in the sea shores (e.g. Cabo Blanco Provincial Reserve) and special features in the steppes (e.g. Meseta Espinosa and El Cordón Provincial Reserve) (Fig. 1b). The area presents a variety of terrestrial ecosystems, classified in five ecological areas (Fig. 1c), dry steppes dominate in the northeast and shrublands and humid steppes in the south, while sub-Andean grasslands, and *Nothofagus* forests and alpine vegetation occupy a narrow fringe in the west (Oliva et al., 2004). *Nothofagus* forest types are distributed from 46° to 52° S, in a decrease range of rainfall (1000–300 mm/year) and elevation (1400–88 m.a.s.l.), while temperature (from 5 to 8 °C) increases from west to east (Veblen et al., 1996) (Fig. 1d).

### 2.2. Potential habitat suitability maps

First, we employed 119 maps (90 × 90 m) of potential habitat suitability belonging to five taxonomic groups, including one mammal (endemic deer), 47 birds, 7 lizards, 10 darkling-beetles and 53 plant species (Appendix 1) (Rosas et al., 2017, 2018, 2019a, 2019b, 2021a; Rosas, 2020). The presence data belong to different sources, highlighting specimens housed (Entomological collection and LJAMM) belong to CONICET, permanent network plots (PEBANPA, Peri et al., 2016b), provincial forest inventory (Peri and Ormaechea, 2013; Peri

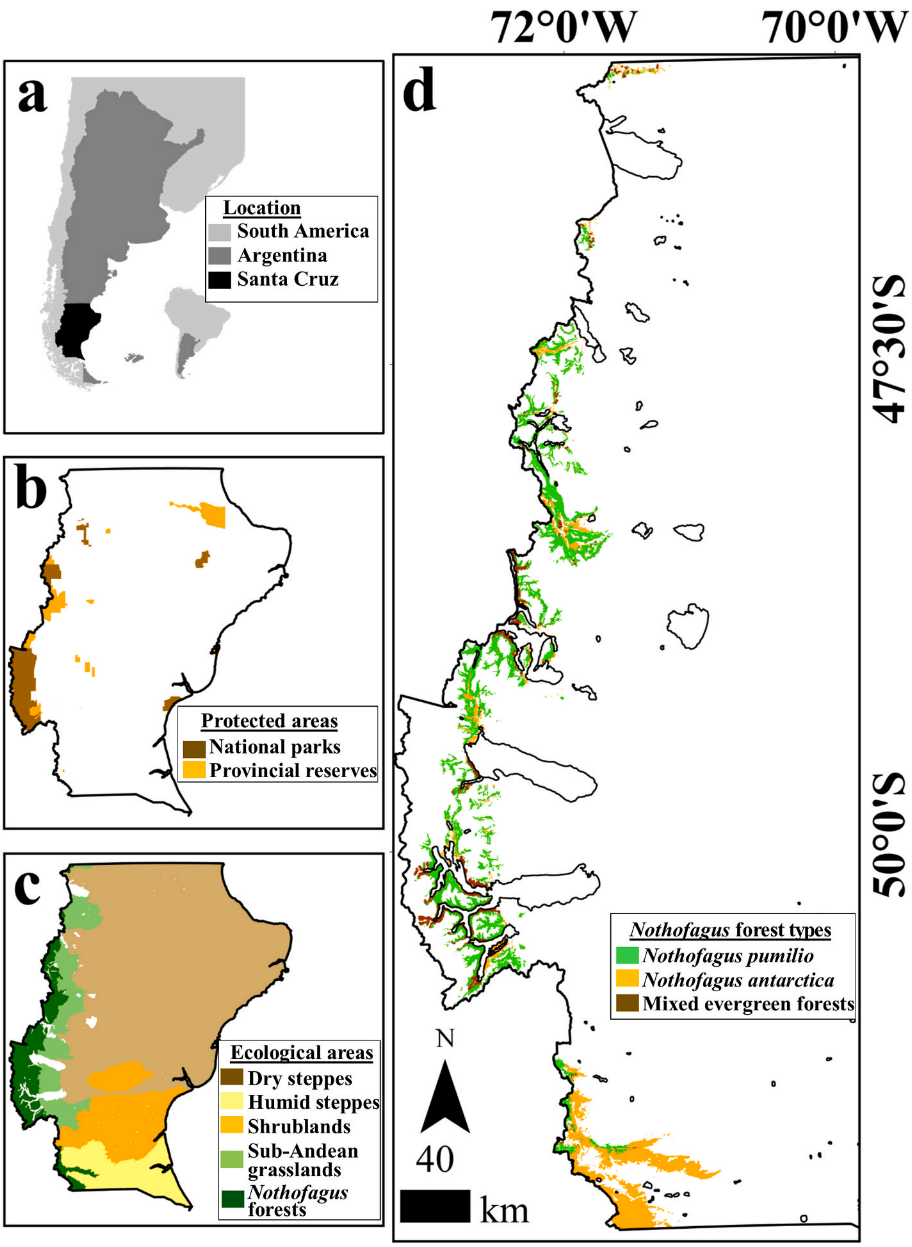


**Fig. 1.** Characterization of the study area: (a) location of Santa Cruz province (black) in Argentina (dark grey); (b) protected areas (brown = national parks, orange = provincial reserves); (c) main ecological areas (brown = dry steppes, yellow = humid steppes, orange = shrublands, light green = sub-Andean grasslands, green = *Nothofagus* forests and alpine vegetation) (modified from Oliva et al., 2004); and (d) *Nothofagus* forests (light green = *N. pumilio*, brown = mixed evergreen forests, orange = *N. antarctica*) (adapted from CIEFAP-MAyDS, 2016). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

et al., 2019), National Administration Park (APN), national (Sistema de Información de Biodiversidad, https://sib.gob.ar) and international online database (https://ebird.org/). The different maps were created following the methodology used in Tierra del Fuego (Martínez Pastur et al., 2016), where models were based on Environmental Niche Factor Analysis (ENFA) (Hirzel et al., 2002) using Biomapper 4.0 software (Hirzel et al., 2004b). ENFA uses the concept of ecological niche (Hutchinson, 1957), and compares environmental variables of locations where the species has been detected with their variability through the whole study area, and predicts the species distribution according to the selected variables (Hirzel et al., 2001). In addition, ENFA calculate two specific ecological relevance indexes: (i) specialization (from 0 to infinite), where higher values indicate that the species tends to live in a narrow range of environment conditions, and (ii) marginality (from 0 to 1), where lower values show that species' requirements do not differ from the average conditions of the study area (Hirzel et al., 2002; Hirzel and Le Lay, 2008; Martínez Pastur et al., 2016). Each model results in a continuous map expressing a range from 0 (minimum) to 100 (maximum potential habitat suitability). Explanatory variables, model outputs and statistical fit analyses were previously described in detail by Rosas et al. (2017, 2018, 2019a, 2019b, 2021a) and Rosas (2020) (Appendix 2, 3 and 4). Each map was visualized into a GIS project (ArcMap 10.0 software, ESRI, 2011) and crossed with a mask based on the NDVI (normalized difference vegetation index) to detect bare soil, ice fields and water bodies (Lillesand and Kiefer, 2000), which were removed from the analyses.

### 2.3. *Maps of potential biodiversity and priority conservation areas considering multi-taxon biodiversity*

Second, we used a cell statistic tool to combine the 119 potential habitat suitability maps (average values for each pixel) considering the different taxonomic groups. We obtained five taxonomic group maps, where four were maps of potential biodiversity that synthetized the information of each taxonomic group (birds, lizards, darkling-beetles and plants) and one potential habitat suitability map for one mammal. Then, we created taxonomic group indexes (GIndex) to weight each taxonomic group map, considering ecological and endemism values

(Appendix 5). For this, we calculated ecological values (ECOi, from 0 to 1) that represents the average of specialization and marginality indexes (ENFA indexes) raised to the third power by species to increase the differences between species, and endemism values (ENDi, Argentina = 1, Patagonia = 5, and Santa Cruz = 10) was a value that include the endemism value of each species, calculated according the available bibliography (Narosky and Yzurieta, 2005; Carrara and Flores, 2013; Breitman et al., 2014, and Flora Argentina http://www.floraargentina.edu.ar). The obtained values were rescaled from 0.5 to 1.0 and then, considering the number of species (N), we calculated the average GIndex for each taxonomic group.

GIndex = (ECOi*ENDi)/N

Third, we elaborated the two final maps using the taxonomic group maps and the indexes (Appendix 5): (i) the map of potential biodiversity (MPB), which was obtained as the combination (sum of values for each pixel) of the weighted five maps considering the taxonomic group indexes, in a GIS project. This map had scores that varied from 1 to 175, so, it was rescaled by a lineal method from 0 to 100; and (ii) the map for priority conservation areas (MPCA), which was obtained using the taxonomic group maps, the indexes and Zonation 4.0 software. Zonation ranks conservation priority over the landscape (from 0 to 1) by iterative removal of the least important pixels, accounting to the total and remaining distributions of species and weights given to species (Moilanen et al., 2005; Di Minin et al., 2014). Two maps were built using basic analyses options in Zonation (Moilanen, 2007). Core-area zonation which bases the ranking considering the most important occurrence of the species in the pixel (e.g. high species endemism) and prioritizes areas with high occurrence levels for narrowly distributed species. In others words, the pixel gets high values if even only one species had a relatively important occurrence in the area. Additive-benefit function which sums values over all species and prioritizes areas with many species overlapping (e.g. high species richness). Finally, in the GIS project, we combined these Zonation maps (average values for each pixel) and we obtained the final MPCA, with scores that varied from 0 to 1.

### 2.4. Data analysis

We analysed the different maps (e.g. environmental variables, taxonomic group maps, MPB, MPCA) using the hexagonal binning processes and considering two spatial scales (provincial and forest landscape matrix) using univariate (ANOVAs) and multivariate analyses. Hexagonal binning processes is a method of aggregating individual data (pixel values) into polygonal regions (Battersby et al., 2017). This spatial methodology can simply and effectively represent complex data sets, improving the ability to analyze and visualize spatial patterns (Briney, 2014). For this, we calculated for each hexagonal area (250 thousand ha for the provincial scale and 5000 ha for the forest landscape matrix scale) the average values of the six environmental variables, the five different taxonomic groups, the potential biodiversity (0–100) and the conservation priority (0–1). Firstly, a principal component analysis was performed to explore hexagon grouping patterns among the different ecological areas (each hexagon = 250 thousand ha, N = from 8 to 77) and forest landscape matrix (each hexagon = 5000 ha), based in two sets of multivariate data: the environmental variables (annual mean temperature, minimum temperature of the coldest month, annual precipitation, precipitation of coldest quarter, elevation and NDVI) (Hijmans et al., 2005; Farr et al., 2007; ORNL DAAC, 2008) and the taxonomic group maps (mammal, birds, lizards, darkling-beetles and plants). For the landscape matrix we considered: (i) three treatments including grasslands (G) (grasslands cover >70%, N = 272), a mix of grasslands and forests (G + Fo) (forest cover between 30% and 50%, N = 70), and forests (Fo) (forest cover >50%, N = 66); and (ii) four treatments including mix of the main forest types and grasslands (*N. pumilio*-G + Fo, *N. antarctica*-G + Fo, N = 52 and 18, respectively) and only the main forest types (*N. pumilio* and *N. antarctica* forests, N = 30 and 36, respectively). Principal component analysis was complemented with a Monte Carlo permutation test (n = 999) to assess the significance of each axis. We selected correlation coefficients among columns to obtain the final cross-product matrices. These analyses were conducted in PCORD 5.0 (McCune and Mefford, 1999). Secondly, one-way ANOVAs were used to assess multi-taxon biodiversity based on the two different spatial analysis (MPB and MPCA) considering different ecological areas, landscape matrix, and several classifications of protected areas (each hexagon = 5000 ha, N > 10) (Fasioli and Díaz, 2011), we used the post-hoc Tukey test ($p < 0.05$) for further mean comparisons. For the landscape matrix we considered: (i) the same three treatments (G, Fo + G and Fo) as was used in principal component analysis, (ii) four treatments including grasslands and different forest types (*N. pumilio*-mixed forests + G, *N. pumilio* forests + G, *N. antarctica* forests + G and *N. antarctica-N. pumilio* forests + G) where treatments included pure and mixed forests, and (iii) one analysis where we excluded the grasslands: *N. pumilio*-mixed forests, *N. pumilio*, *N. antarctica-N. pumilio* and *N. antarctica* forests. Further, because weight can influence the final maps, we conducted a sensitivity analysis where taxonomic maps presented the same weight (GIndex = 1) and calculated the change in MPB and MPCA. These analyses were performed in Statgraphics software (Statistical Graphics Corp., USA).

## 3. Results

### 3.1. Taxonomic group maps and environmental variables

The potential habitat suitability maps of each species showed different habitat requirements (e.g. marginality and specialization index) (Fig. 2 and Appendix 6). Lizards (blue colour) and darkling-beetles (brown colour) presented a narrow range of marginality values, while birds (orange colour) and plants (green colour) presented a large range of marginality and specialization values. Considering each group (without the mammal), lizards had the highest specialization (7.9) and birds the lowest (3.4), while plants presented the highest average marginality (1.5) and darkling-beetles the lowest (0.8). At species level, *Cinclodes patagonicus* (bird) showed the lowest specialization value (1.3), while *Berberis empetrifolia* (plant) one of the highest specialization values (13.1), *Mulguraea tridens* (plant) the lowest marginality (0.5), and the mammal *Hippocamelus bisulcus* (grey colour) the highest marginality (4.9).

Principal component analysis showed different ordination patterns of ecological areas, vegetation types, and forested landscape types, depending on the variables used to characterize the hexagons (e.g. environmental variables or taxonomic group maps) (Fig. 3 and Table 1). At provincial level, we observed that hexagons of ecological areas in the east and west were separated with both sets of multivariate information

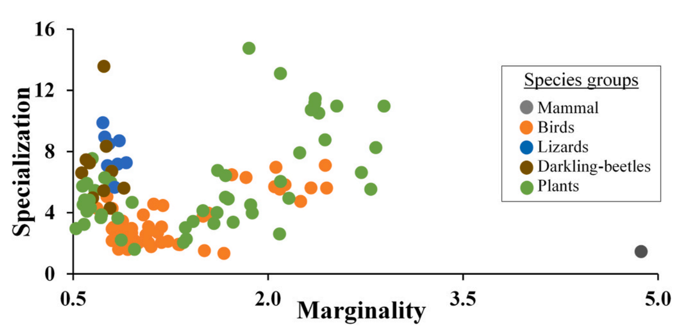


**Fig. 2.** Specialization vs. marginality indexes classified according to the studied taxonomic groups. Where: mammal = grey, birds = orange, lizards = blue, darkling-beetles = brown, and plants = green. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

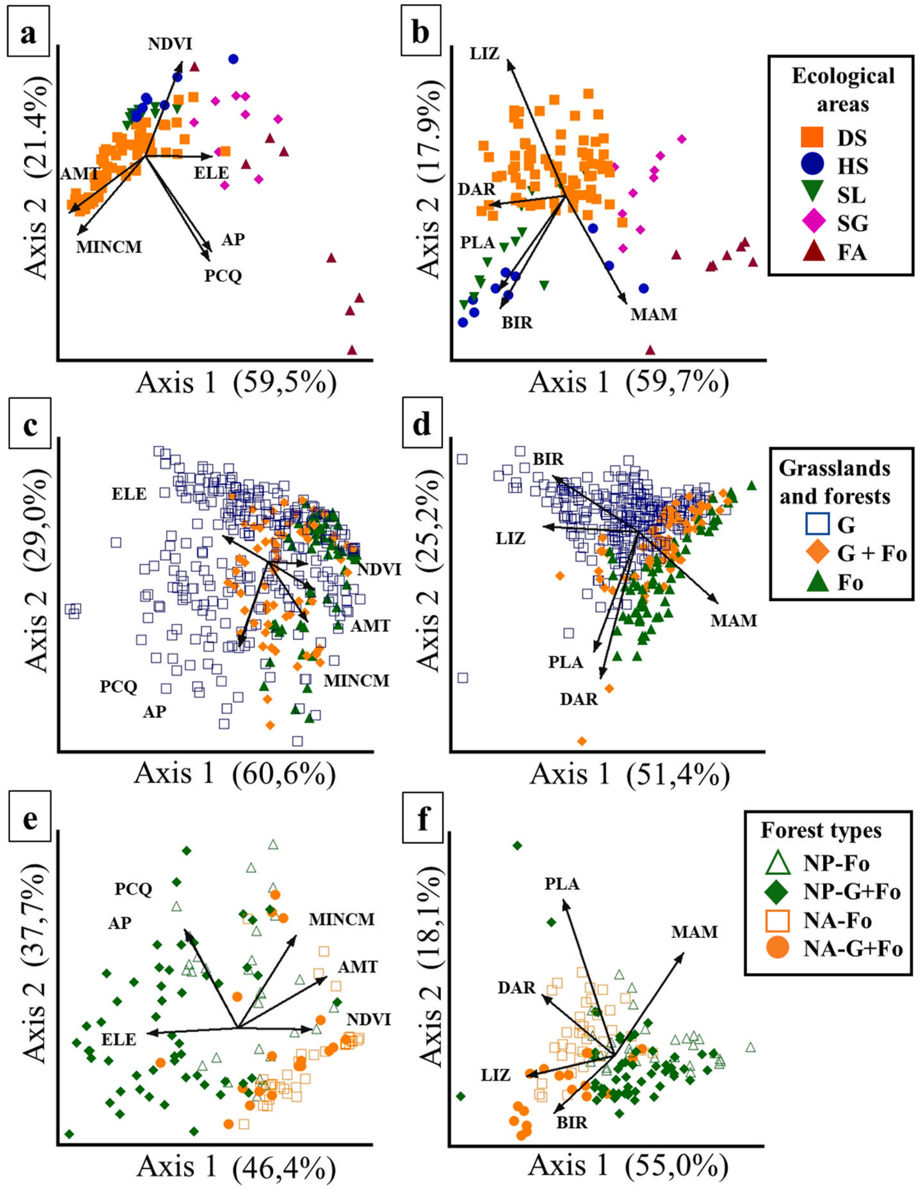


**Fig. 3.** Hexagon ordination by principal component analysis using environmental variables (left) and habitat suitability/potential biodiversity of taxonomic groups (right), considering for classification the characteristics of each hexagon, according to: (a, b) ecological areas (DS = dry steppe, HS = humid steppe, SL = shrublands, SG = sub-Andean grasslands, FA = forests and alpine vegetation); (c, d) vegetation types (G = grasslands, G + Fo = grasslands and forests, Fo = forests), and (e, f) vegetation and forest types (NP-Fo = *Nothofagus pumilio* forests, NP-G + Fo = *N. pumilio* forests and grasslands, NA-Fo = *N. antarctica* forests, NA-G + Fo = *N. antarctica* forests and grasslands). Where: AMT = annual mean temperature, MINCM = min temperature of coldest month, AP = annual precipitation, PCQ = precipitation of coldest quarter, ELE = elevation, NDVI = normalized difference vegetation index, MAM = mammal, BIR = birds, LIZ = lizards, DAR = darkling-beetles, PLA = plants. Percentages in the axes show the proportion of explained variance.

(taxonomic groups and environmental variables). However, when environmental variables were used alone, hexagons of different steppes areas were strongly overlapped, and forest and alpine vegetation showed a major dispersion (Fig. 3a). Temperature variables (annual mean temperature and precipitation of the coldest quarter) were associated to dry steppe and NDVI to humid steppes and shrublands, while precipitation variables (annual precipitation and precipitation of coldest quarter) and elevation were related to western ecological areas (sub-Andean grasslands and forests and alpine vegetation). On the other hand, when taxonomic groups information was used alone, grouping of hexagons according to the different ecological areas have presented more dispersion for dry steppe, humid steppe and shrublands, while forest and alpine vegetation was less dispersed. Moreover, we observed that lizards and darkling-beetles were associated to hexagons of dry steppe, plants and birds to shrublands and humid steppe, while the mammal was related to sub-Andean grasslands and forests and alpine vegetation (Fig. 3b).

At forest landscape level, we observed that hexagons of all categories were overlapped with both sets of multivariate information (Fig. 3c and d), although the grasslands showed a great dispersion when environmental variables were used (Fig. 3c). Temperature and NDVI variables were slightly more associated to hexagons of forests (Fo) and the combination of grasslands and forests (G + Fo), while precipitation variables and elevation were orthogonal but slightly related to grasslands (G) (Fig. 3c). On the other hand, we observed that birds and lizards were associated to more dispersed hexagons of G, plants and darkling-beetles to G + Fo, while the studied mammal was related to Fo (Fig. 3d). In addition, principal component analysis showed that hexagons of *N. pumilio*-G + Fo are mostly split from the other categories, while *N. pumilio* and *N. anarctica* forests were also separated, where *N. pumilio*-G + Fo presented a great dispersion when environmental variables were analysed (Fig. 3e). Low values of precipitation variables were associated with *N. anarctica* forests, and high elevations were related to *N. pumilio*-G + Fo, while temperature and NDVI variables were also associated to *N. anarctica* forests, as well as to *N. antarctica*-G + Fo (Fig. 3e). On the other hand, when taxonomic group information was used, hexagons were much more overlapped (Fig. 3f), where darkling-beetles and plants were positively related to *N. anarctica* forests, and lizards and birds were associated to *N. antarctica*-G + Fo (with lizards opposited to *N. pumilio* forests), while the mammal was related to *N. pumilio* forests (Fig. 3f).

**Table 1**
Principal component analyses (PCA) statistical results for environmental variables and habitat suitability/potential biodiversity of taxonomic groups according to ecological areas, vegetation types and vegetation and forest types. Where: AMT = annual mean temperature, MINCM = min temperature of coldest month, AP = annual precipitation, PCQ = precipitation of coldest quarter, ELE = elevation, NDVI = normalized difference vegetation index, MAM = mammal, BIR = birds, LIZ = lizards, DAR = darkling-beetles, PLA = plants.

| Analysis factor | Axis | Eigenvalue | Explained variance | p-value | Axis | Eigenvalue | Explained variance | p-value |
|---|---|---|---|---|---|---|---|---|
| Ecological areas | 1 | 3.572 | 59.5% | 0.001 | 1 | 2.985 | 59.7% | 0.001 |
| | 2 | 1.282 | 21.4% | 0.012 | 2 | 0.894 | 17.9% | 1.000 |
| | | | **Eigenvector** | | | | **Eigenvector** | |
| | **Environmental variables** | | **1** | **2** | **Taxonomic group** | | **1** | **2** |
| | AMT | | −0.48 | −0.28 | MAM | | 0.41 | −0.47 |
| | MINCM | | −0.43 | −0.40 | LIZ | | −0.39 | 0.59 |
| | AP | | 0.42 | −0.49 | PLA | | −0.45 | −0.41 |
| | PCQ | | 0.40 | −0.54 | DAR | | −0.52 | −0.04 |
| | ELE | | 0.43 | −0.00 | BIR | | −0.44 | −0.49 |
| | NDVI | | 0.23 | 0.48 | | | | |
| **Analysis factor** | **Axis** | **Eigenvalue** | **Explained variance** | **p-value** | **Axis** | **Eigenvalue** | **Explained variance** | **p-value** |
| Vegetation types | 1 | 3.634 | 60.6% | 0.001 | 1 | 2.569 | 51.4% | 0.001 |
| | 2 | 1.744 | 29.0% | 0.001 | 2 | 1.259 | 25.2% | 0.001 |
| | | | **Eigenvector** | | | | **Eigenvector** | |
| | **Environmental variables** | | **1** | **2** | **Taxonomic group** | | **1** | **2** |
| | AMT | | 0.49 | −0.20 | MAM | | 0.49 | −0.34 |
| | MINCM | | 0.41 | −0.43 | LIZ | | −0.59 | 0.02 |
| | AP | | −0.31 | −0.60 | PLA | | −0.27 | −0.57 |
| | PCQ | | −0.31 | −0.61 | DAR | | −0.23 | −0.70 |
| | ELE | | −0.48 | 0.18 | BIR | | −0.53 | 0.26 |
| | NDVI | | 0.41 | −0.01 | | | | |
| **Analysis factor** | **Axis** | **Eigenvalue** | **Explained variance** | **p-value** | **Axis** | **Eigenvalue** | **Explained variance** | **p-value** |
| Vegetation and forest types | 1 | 2.786 | 46.4% | 0.001 | 1 | 2.753 | 55.0% | 0.001 |
| | 2 | 2.264 | 37.7% | 0.001 | 2 | 0.903 | 18.1% | 0.001 |
| | | | **Eigenvector** | | | | **Eigenvector** | |
| | **Environmental variables** | | **1** | **2** | **Taxonomic group** | | **1** | **2** |
| | AMT | | 0.51 | 0.29 | MAM | | 0.44 | 0.50 |
| | MINCM | | 0.33 | 0.53 | LIZ | | −0.57 | −0.10 |
| | AP | | −0.30 | 0.56 | PLA | | −0.33 | 0.76 |
| | PCQ | | −0.30 | 0.56 | DAR | | −0.47 | 0.29 |
| | ELE | | −0.52 | −0.03 | BIR | | −0.39 | −0.30 |
| | NDVI | | 0.42 | −0.01 | | | | |

### *3.2. Multi-taxon biodiversity assessment*

The five taxonomic group maps changed through the landscape (Fig. 4). The mammal presented the narrowest distribution in the west near to the forests, while birds and plants presented a wide distribution around the province and lizards and darkling-beetles presented a narrower distribution in the east. Particularly, the mammal presented a discontinuous distribution with low potential habitat suitability values in the north and south, while the highest values were found in the central-west area of the province (near to the biggest lakes) (Fig. 4a). Birds and plants presented different areas with high potential biodiversity values, the highest values of birds group were located in the south-east, the lowest values in the central and medium values in the west (Fig. 4b), while plants group presented the highest values in the east and in the extreme south-east of the province (Fig. 4e). On the other hand, lizards presented high potential biodiversity values in the north-east and medium values in the central-north and south of the province (Fig. 4c), while darkling-beetles have the highest values in the south-east and medium values in the north (Fig. 4d).

The combination of the five taxonomic group maps allowed us to obtain the final maps to assess multi-taxon biodiversity (MPB and MPCA) for Santa Cruz province (Fig. 5). The MPB presented low (<30) values near to the forests and ice fields in the west, while high (>80) values occurred in the east. However, values increased (from 40 to 70) near to the big lakes and forests in the southwest (Fig. 5a). Moreover, MPCA presented similar patterns through the landscapes to those observed in MPB, but with higher (>0.8) values located in western areas (Fig. 5b).

ANOVAs showed that potential biodiversity and conservation priority values changed according to the different ecological areas and forest landscape matrix (Table 2). At provincial level, both maps showed significant differences, with higher values in shrublands and humid steppe, following by dry steppe areas that presented medium values. MPB ($F = 17.95$, $p < 0.001$) showed low values in forests and alpine vegetation, while MPCA ($F = 9.30$, $p < 0.001$) presented the highest value for this ecological area. In addition, both maps presented the lowest values in sub-Andean grasslands. At forest landscape matrix, ANOVAs showed that maps presented significant differences when grasslands and forests were compared ($F = 133.35$, $p < 0.001$), where Fo presented the highest values, followed by Fo + G, and G, which presented the lowest values. In addition, ANOVAs showed that maps presented significant differences when grasslands and the different forests types were considered ($F = 3.70$, $p < 0.016$), where the highest values occurred when *N. pumilio* and *N. antarctica* forests were combined with grasslands landscapes. Finally, ANOVAs showed that MPB presented significant differences when only the forests types were considered ($F = 17.65$, $p < 0.001$) highlighting *N. antarctica* forests which presented the highest values, while MPCA did not show significant differences ($F = 0.41$, $p = 0.747$). In addition, sensitivity analysis (Appendix 7) showed that the weight (GIndex) did not influence potential biodiversity and conservation priority values at provincial level. However, at forest landscape matrix Tukey test showed less differences among grasslands and forest types, where the highest potential biodiversity values occurred in two types, while the highest conservation priority values occurred in three types.

In addition, ANOVAs showed that maps changed when different classifications of protected areas were evaluated (Table 3). While the higher potential biodiversity values occurred outside of protected areas, the higher priority conservation values occurred inside. In fact, national parks presented the highest conservation priority values ($F = 23.05$, $p <$

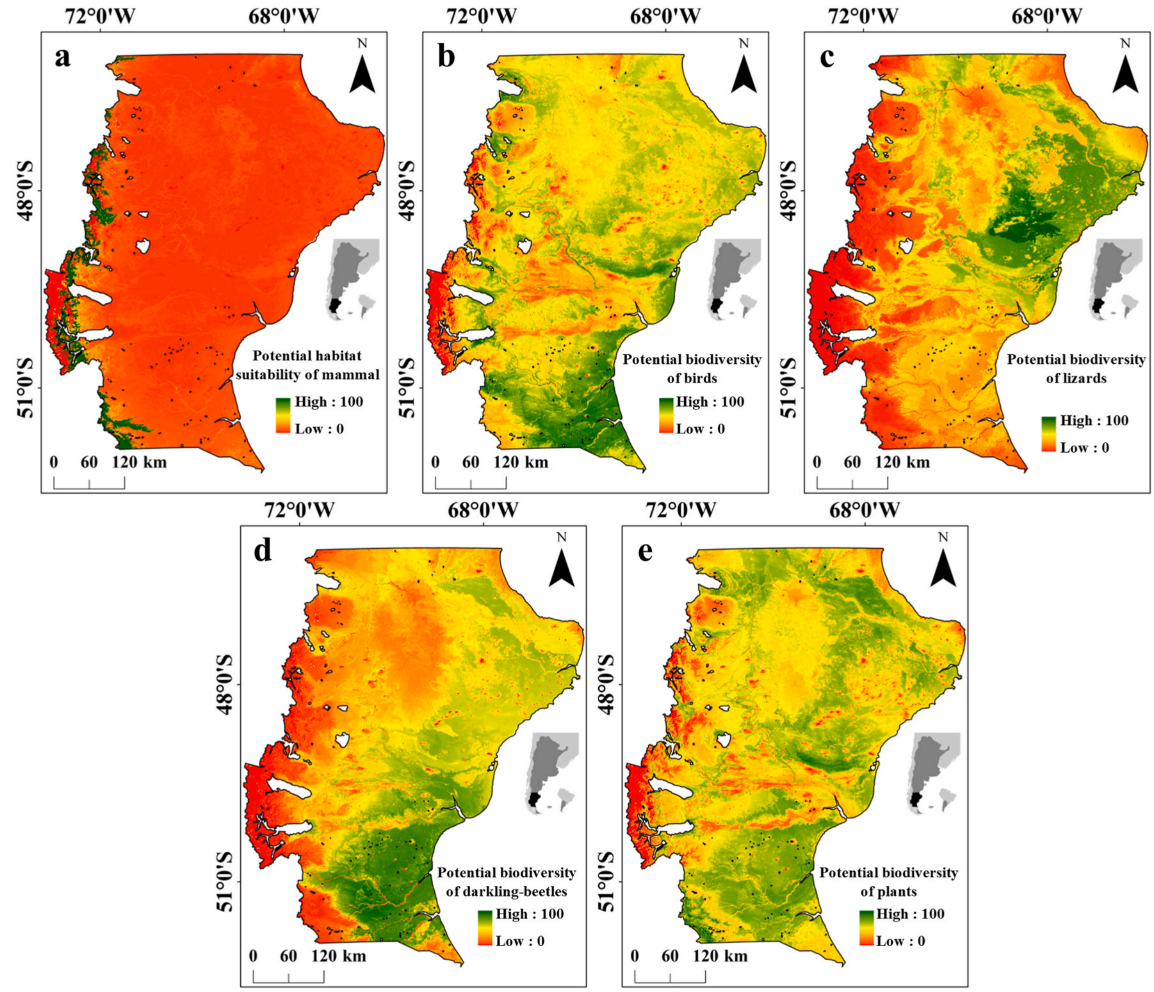


**Fig. 4.** Maps of taxonomic groups in Santa Cruz province, where red colour indicates low values (0) and green colour indicates high values (100). Potential habitat suitability of mammal (a), potential biodiversity of birds (b), potential biodiversity of lizards (c), potential biodiversity of darkling-beetles (d) and potential biodiversity of plants (e). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

0.001), while provincial reserves showed the highest potential biodiversity values ($F = 21.72$, $p < 0.001$). The main national parks and provincial reserves presented also significant differences among them, where Monte León National Park, and Meseta Espinosa and El Cordón Provincial Reserve were those with the highest potential biodiversity values within each analysis, and Los Glaciares National Park, Peninsula de Magallanes, Lago del Desierto and Tucu-Tucu Provincial Reserves showed the highest conservation priority values within each analysis. The sensitivity analysis (Appendix 8) showed that the weight (GIndex) did not influence potential biodiversity and conservation priority values when protection (protected and unprotected) and protection types (national park and provincial reserves) are considering. However, when different national parks were considered Tukey test showed that potential biodiversity values did not change, while two national parks (Los Glaciares and Monte Leon) presented the highest conservation priority values. In addition, Tukey test showed that potential biodiversity values did not change when provincial reserves were considered, however the highest values increased (e.g. Meseta Espinosa and El Cordón from 68.84 to 73.44), while conservation priority values decreased (e.g. Lago del Desierto from 0.81 to 0.76).

## 4. Discussion

### 4.1. Taxonomic group maps

Usually, biodiversity conservation strategies in remote areas such as Southern Patagonia, only included specific (e.g. endangered or charismatic species) (Rosas et al., 2017) or taxonomic target groups (e.g. lizards) (Breitman et al., 2014, 2015). Despite the relevant biological information that species distribution models (e.g. potential habitat suitability and environmental requirements) provide at a low cost compared with field surveys in large regions (e.g. Santa Cruz landscapes included 243,943 km$^2$), few studies considered this type of methodologies (Tulloch et al., 2016) for the development of conservation strategies at a landscape level. For example, national park administration of Argentina provided a framework to support conservation strategies using expert knowledge on species distribution, based on general information (e.g. occurrence points near routes and draw coarse species range maps) for Patagonian steppes (Chehébar et al., 2013). During the last years, new spatial information of potential habitat suitability in Southern Patagonia (Rosas et al., 2017, 2018, 2019a, 2019b, 2021a; Rosas, 2020) allowed to design an improved framework, considering species presence data of different taxonomic groups and open-access environmental variables in the web to support conservation strategies. Different studies showed the advantages of this type of maps compared with general distribution maps and the potential to combine multiple

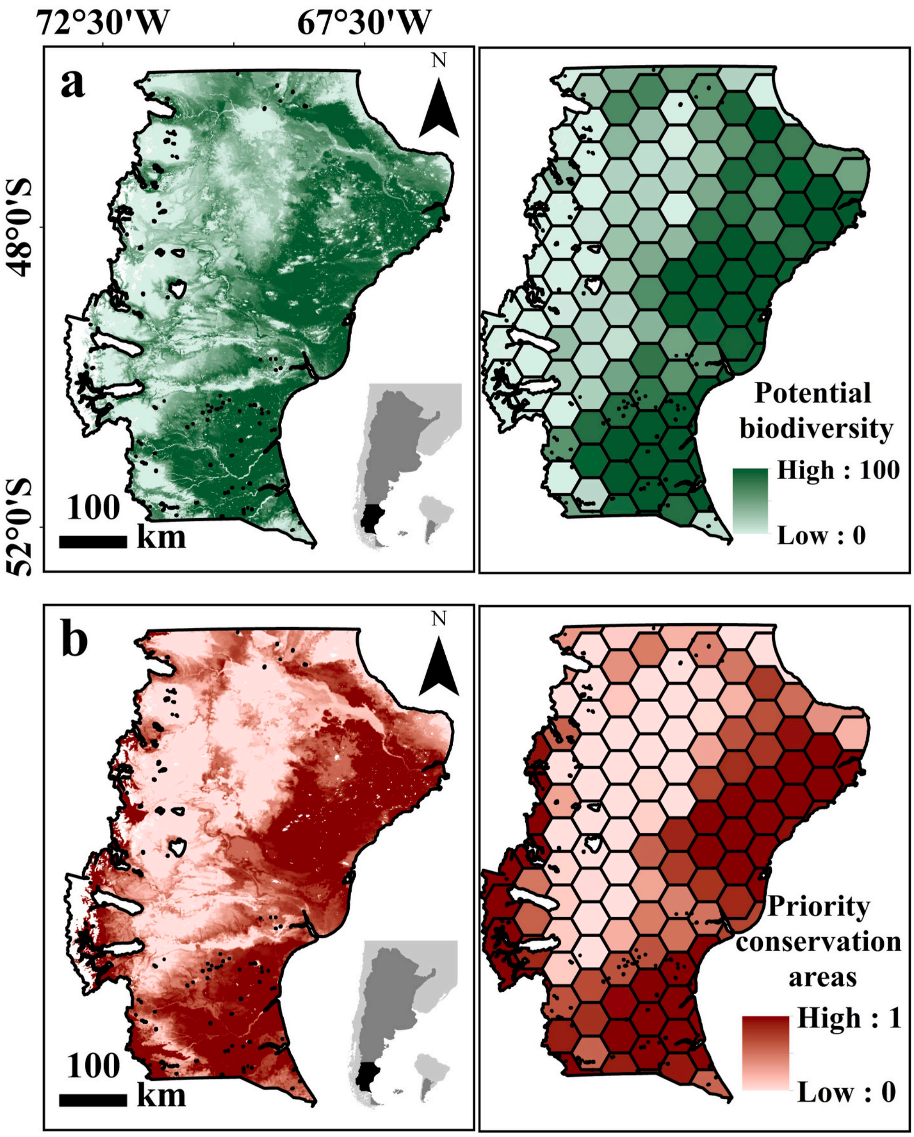


**Fig. 5.** Maps of potential biodiversity (a) and priority conservation areas (b) of selected taxonomic groups species in Santa Cruz province (left) in hexagons of 250 thousand ha obtained through the hexagonal binning process (right). High intensity colours represent higher values and light colours represent lower values. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

species to improve conservation planning (Tulloch et al., 2016; Sofaer et al., 2019).

Our analyses suggest that, *Hippocamelus bisulcus* (mammal) occurred in the most extreme environmental condition (high marginality), while *Berberis empetrifolia* (plant) presented the narrowest distribution (high specialization). In fact, different taxonomic groups (e.g. lizards and darkling-beetles) presented lower marginality values in similar environmental condition than the other groups (e.g. plants and birds) (Rosas, 2020). The principal component analysis confirmed these results at regional and at forest landscape matrix levels. The studied mammal was more related to ecological areas in the west (sub-alpine grasslands and forest and alpine vegetation) and was mainly associated with elevation and precipitation. Although some studies indicated that this mammal is a forest related species (Corti et al., 2011; Quevedo et al., 2017), another study showed that the optimal habitat suitability values were associated to ecotone (grassland and forests) in the western areas of the province (Rosas et al., 2017). Here, *Hippocamelus bisulcus* was related to *N. pumilio* forests and ecotone areas when landscape matrix was considered in the principal component analysis.

Contrarily, the highest potential biodiversity values of lizards and darkling-beetles were related to dry steppes where different temperature regimes occurred. In this sense, different studies indicate the strong relationship among these steppe species and temperature regimes. Fernández et al. (2011) showed high performance in a wide range of low temperature conditions for some lizard species (e.g. *Liolaemus sarmientoi*), which was supported by Rosas et al. (2018) who identified the highest values at low temperature values in the southern areas. In addition, darkling-beetles live in a wide range of temperature conditions, where some studies identified high adaptations (e.g. morphological and behaviour) and ecological plasticity for desert areas (Matthews et al., 2010). Rosas et al. (2019a) identified optimal values at high temperature values for some species (e.g. *Nyctelia fitzroyi*) and in low temperatures for others (e.g. *Praocis bicarinata*). Although most of these species are related to dry steppe areas (Carrara and Flores, 2013; Breitman et al., 2015), the highest potential biodiversity values of darkling-beetles were found in humid steppes (Rosas et al., 2019a), where our principal component analysis showed a slight association with this ecological area. However, at a landscape matrix level showed that the highest values of darkling-beetles were related to *N. antarctica* forests, while other studies identify areas of micro-endemism in steppe

**Table 2**
One-way ANOVAs of potential biodiversity (obtained through the maps of potential biodiversity-MPB) and conservation priority (obtained through the maps of priority conservation areas-MPCA) for the studied taxonomic groups in Santa Cruz province, comparing the dominant characteristic of hexagons: ecological areas (DS = dry steppe, HS = humid steppe, SL = shrublands, SG = sub-Andean grasslands, FA = forests and alpine vegetation) and the forest landscape matrix (G = grasslands, Fo + G = forests and grasslands, Fo = forests, NP-MIX + G = *Nothofagus pumilio,* mixed forests and grasslands, NP + G = *N. pumilio* forests and grasslands, NA + G = *N. antarctica* forests and grasslands, NP-NA + G = *N. pumilio, N. antarctica* forests and grasslands, NP-MIX = *N. pumilio* and mixed forests, NP = *N. pumilio*, NA-NP = *N. antarctica* and *N. pumilio* forests, and NA = *N. antarctica* forests).

| Analysis factor | | | MPB | MPCA |
|---|---|---|---|---|
| Ecological areas | | DS | 57.39 b | 0.47 ab |
| | | HS | 63.77 bc | 0.70 c |
| | | SL | 66.66 c | 0.63 bc |
| | | SG | 35.32 a | 0.37 a |
| | | FA | 43.61 a | 0.79 c |
| | | F(p) | 17.95 (<0.001) | 9.30 (<0.001) |
| Forest landscape matrix | Grasslands and forests | G | 39.70 a | 0.55 a |
| | | Fo + G | 48.19 b | 0.83 b |
| | | Fo | 54.89 c | 0.93 c |
| | | F(p) | 174.46 (<0.001) | 133.35 (<0.001) |
| | Grasslands and forest types | NP-MIX + G | 45.22 a | 0.77 a |
| | | NP + G | 46.00 a | 0.79 ab |
| | | NA + G | 49.95 ab | 0.83 ab |
| | | NA-NP + G | 52.62 b | 0.90 b |
| | | F(p) | 9.96 (<0.001) | 3.70 (0.016) |
| | Forest types | NP-MIX | 46.06 ab | 0.93 |
| | | NP | 49.02 a | 0.93 |
| | | NA-NP | 54.09 b | 0.95 |
| | | NA | 59.90 c | 0.93 |
| | | F(p) | 17.65 (<0.001) | 0.41 (0.747) |

F(p): F test and associated probability. Different letters (a, b and c) in the same column and factor showed differences in mean comparisons by Tukey test (p < 0.05).

**Table 3**
One-way ANOVAs of potential biodiversity (obtained through the maps of potential biodiversity-MPB) and conservation priority (obtained through the maps of priority conservation areas-MPCA) for the studied taxonomic groups in Santa Cruz province considering different classifications of protected areas.

| Analysis factor | | MPB | MPCA |
|---|---|---|---|
| Protection | Protected | 48.59 a | 0.67 b |
| | Unprotected | 56.14 b | 0.51 a |
| | F(p) | 85.00 (<0.001) | 155.91 (<0.001) |
| Types | National Park | 45.20 a | 0.72 b |
| | Provincial Reserves | 52.45 b | 0.61 a |
| | F(p) | 21.72 (<0.001) | 23.05 (<0.001) |
| National parks | Los Glaciares | 40.40 a | 0.76 b |
| | Perito Moreno | 41.11 a | 0.66 a |
| | Bosques Petrificados de Jaramillo | 59.82 b | 0.60 a |
| | Monte León | 73.69 c | 0.73 ab |
| | F(p) | 117.72 (<0.001) | 6.08 (<0.001) |
| Provincial reserves | San Lorenzo | 29.89 a | 0.52 abc |
| | Lago del Desierto | 41.87 ab | 0.81 d |
| | Bosque Petrificado. Ea. La Urbana | 43.44 ab | 0.29 a |
| | Tucu-Tucu | 43.45 b | 0.74 cd |
| | Península de Magallanes | 43.96 b | 0.87 d |
| | Meseta Espinosa and El Cordón | 68.84 c | 0.58 b |
| | F(p) | 53.46 (<0.001) | 15.27 (<0.001) |

F(p): F test and associated probability. Different letters (a, b, c and d) in the same column and factor showed differences in mean comparisons by Tukey test (p < 0.05).

areas near to these forests (Carrara and Flores, 2013). In addition, the highest values for lizards were associated to ecotone areas (*N. antarctica*-G + Fo), where Breitman et al. (2012) identified hypothetical refuge areas for the *Liolaemus lineomaculatus* group. However, Breitman et al. (2014) and Rosas et al. (2018) showed lower richness and potential biodiversity values in areas close to forests.

Plant and bird species occurred in a large range of environmental conditions, where the highest potential biodiversity values were mostly related to humid steppes and shrublands associated with higher NDVI values. Several studies identified strong functional relationships between both taxonomic groups (Kissling et al., 2008), e.g. Paiaro et al. (2017) found some bird species (e.g. *Zonotrichia capensis*) associated to endemic plant species (e.g. *Anarthrophyllum desideratum*) in the Patagonian steppes. However, other plant and bird species (e.g. *Blechnum penna-marina* and *Aphrastura spinicauda*, respectively) have been specifically associated to forests (e.g. mixed evergreen forest) (Martínez Pastur et al., 2015; Benitez et al., 2019) in Tierra del Fuego and Isla de los Estados, Argentina. In fact, some studies identified plant species associated with different forest types, e.g. *Viola magellanica* was related to *N. pumilio* forests, *Carex andina* was associated with *N. antarctica* forests in Santa Cruz province (Roig, 1998; Rosas et al., 2019b). Our results at the landscape level showed that the highest values of biodiversity for plants were related to *N. antarctica* forests, where more shrub and grass species abundantly grown (Peri and Ormaechea, 2013; Peri et al., 2016a) and where more hot-spot areas for these species have been identified (Rosas et al., 2019b). In addition, the higher values for birds were associated to ecotone areas (*N. antarctica*-G + Fo). In this sense, Altamirano et al. (2020) reported that habitat structural heterogeneity (e.g. old-growth montane forests and subalpine environments) was positively associated with bird diversity in the southern temperate Chilean Andes.

### 4.2. Multi-taxon biodiversity assessment: scale level analysis

The different ecological requirements of each taxonomic groups showed the needs to combine all maps in a single map to complement each species maps to improve conservation multi-taxon biodiversity strategies at several spatial scales (Lehtomäki et al., 2009; Khosravi et al., 2019; Miu et al., 2020). There are different ways to combine distribution maps considering different conservation targets (e.g. umbrella, endangered, keystone, endemic or focal species) (Gloves et al., 2002). Usually, conservation actions focused on extinction risk of species at local scales, and conservation networks of protected areas focused at regional scales (Wiens and Bachelet, 2010). In this sense, we used ecological indexes (ECOi) and endemism information (ENDi) to emphasize specific species of each taxonomic groups. For example, endangered species (e.g. *Hippocamelus bisulcus*, ECOi*ENDi = 1,00) (Black-Decima et al., 2016), species living in a narrow distribution (e.g. Nyctelia fitzroyi, ECOi*ENDi = 0.88) (Breitman et al., 2015), related to extreme climatic condition (*Liolaemus sarmientoi*, ECOi*ENDi = 0.76) (Fernández et al., 2011) or associated to specific forest types (*Escallonia rubra* and *Pygarrhichas albogularis*, ECOi*ENDi = 0.77 and 0.58) (McGehee and Eitniear, 2007; Rosas et al., 2019b). Despise the GIndex influence the final maps, our analyses suggested that the selection of the spatial decision-support tool depends on the scale of analysis (e.g. provincial or regional) and on the analysed features (e.g. ecological areas or forest landscape matrix). At provincial level, both maps showed similar priority conservation areas with higher values in the shrublands and humid steppes. Both methodologies highlighted the importance of richness, where several species presented high potential habitat suitability values in these areas (Rosas et al., 2018, 2019a; Rosas, 2020). In addition, MPCA also identified the highest values in forests and alpine vegetation ecological area, where high potential habitat suitability values of endemic and endangered species (e.g. *Hippocamelus bisulcus*) (Rosas et al., 2017) determined the conservation priority values. Despite the relevant importance of biodiversity in Patagonian steppe areas (Chehébar et al., 2013; Peri et al., 2013), main economic interest (e.g. livestock, agriculture) slows down most of the conservation strategies (Venter et al., 2018). In contrast, most of Patagonian native forests are under strict protection (Martín and Chehébar, 2001), mainly with the goal to preserve natural mountain landscapes (Catalan et al., 2017).

When the landscape matrix level was analysed, values of both methodologies increasing with forest cover (>50% forest cover) and with ecotone areas where different forest types are combined (e.g. *N. antarctica-N. pumilio* forests + G). Forest ecotone areas favoured biodiversity hotspots (e.g. vascular plants, epiphytic lichens, soil macro-arthropods) according to several studies (Hauck et al., 2014; Sottile et al., 2015). In the same study area, Rosas et al. (2019b) found high potential biodiversity values of understory plant species associated to environmental heterogeneity (e.g. open-lands), which support a major plant diversity compared to other natural environments (Gargaglione et al., 2014). In addition, Altamirano et al. (2020) identified ecotonal temperate forest areas as relevant for avian species richness and the associated functional diversity. The complex dynamic of the ecotone areas determines that disturbance events can produce rapid or abrupt changes by increasing (e.g. fire events) or decreasing (e.g. grazing) plant diversity (Sottile et al., 2015). In addition, only MPB methodology identified significant differences among the different forest types, e.g. *N. antarctica* forests presented the highest potential biodiversity values that must be considered for conservation on unprotected areas (e.g. private lands), where new economic management (e.g. silvopastoral strategies) must been implemented to improve *in situ* conservation (land-sharing strategy) (Peri et al., 2016a).

### 4.3. Spatial decision-support maps: how to improve the multi-taxon biodiversity conservation?

Wiens and Bachelet (2010) suggested that efforts of conservation should be focused on a subset of species that characterize the area and contributes to maintain the entire biodiversity through direct (e.g. designing strict conservation areas) or indirect actions (e.g. reducing threats such as hunting or livestock). Our results indicate the importance to define the conservation strategy at different scale levels (e.g. richness or endangered species). Thus, while MPCA highlighted forested areas at provincial level, MPB highlighted *N. antarctica* forests at the landscape matrix level. The role of protected areas was improved in Argentina during the last years (Martín and Chehébar, 2001), however, the protected areas network must include lands such as steppes and shrublands mainly used for ranching (Watson et al., 2014; Venter et al., 2018). Many species need protection (e.g. megafauna or several bird species with specific habitat specifications), requiring large conservation areas where the integration of design rules and ecosystem management approaches at the species, ecosystem, and landscape levels must be included (e.g. provision of ecosystem services) (Bottrill and Pressey, 2012; Adams et al., 2019). Our results indicate that the effectivity of the current network of protected natural areas depend on the considered methodology, where MPB showed the highest values on unprotected areas and MPCA on protected areas. Thus, while MPB exposed the need to incorporate new areas to protect higher values of potential biodiversity at both scale levels, MPCA indicated similar results, but to protect forested areas. In fact, both methodologies highlighting existing protected areas, e.g. Monte León (located in steppe areas) using MPB, and Los Glaciares (located in forested areas) using MPCA.

Conservation planning has improved worldwide by increasing the availability of biological and environmental open-access databases (e.g. WorldClim), GIS-based tools, mathematical algorithms, and software (e.g. Marxan and Zonation) (Moilanen et al., 2011; Daigle et al., 2020). All these inputs and tools give feasible solutions to complex conservation problems (Thomassen et al., 2011; Tulloch et al., 2016), such as maps of conservation priority areas for planning framework (Bottrill and Pressey, 2012). However, the successful transition between regional-scale plans and local-scale actions needs a better understanding among the different factors (Adams et al., 2019). Despite meaning of the natural protected areas has changed during the last years (Watson et al., 2014), new protected areas are not targeting places with high conservation interest (e.g. threatened vertebrate species) and are located in areas that minimize conflicts with other land uses (e.g. agriculture) (Venter et al., 2018). This declaration is consistent with our results, where areas with high biodiversity values (considering the overlapping of different maps and the single MPB) and economic land uses (e.g. humid steppes and *N. antarctica* forests) are not included in the natural protected networking (e.g. old network and new ones). In this context, the most common conservation approach in Argentina (e.g. Santa Cruz province) were private donations to National Park Administration to increase conservation areas. For example, Ea. El Rincón in Perito Moreno National Park (N° 641/16), Piedra del Fraile in Los Glaciares National Park (N° 327/19), and Reservas Silvestres in Patagonia National Park (N° 838/18 and N° 326/19). In this context, these maps can allow us to evaluate these conservation actions and detect potential conservation gaps. Furthermore, some high mountain protected areas have assumed that their natural condition would not change over time or will recover after human impacts (Catalan et al., 2017). Martín and Chehébar (2001) indicated that most of the natural protected areas in the Andes mountains were chosen due to its location than its ecological values (e.g. geopolitical borders). Mountain study areas presented low potential biodiversity values (Rosas et al., 2018, 2019a), however, some endangered species (e.g. *Hippocamelus bisulcus*) present critical areas that need to be considered for conservation (Corti et al., 2011; Rosas et al., 2017).

In addition, it is necessary to define conservation planning under climate change scenarios, which can modify the habitat suitability, the species interactions, and also, the community assemblages (Reside et al., 2018). During the last years, new and innovative conservation approaches are recommended under competing land uses and climate change scenarios: (i) land-sharing and ecological corridors in managed areas can increase the conservation efforts by combining a mosaic of land-use types, e.g. buffer zones around protected areas, agroforestry, and more grazing sustainable strategies (Gillson et al., 2013; Crespin and Simonetti, 2019). (ii) Moveable protected areas by considering the dynamic in protection for a specific time (e.g. reproduction period or refuge for extreme events) and/or space (Carvalho et al., 2011). (iii) New protected areas by considering species distribution models with future climate projections of potential habitat suitability (Kujala et al., 2013). (iv) Restoration by including degraded systems or national parks, and creating corridors to expand actual natural protected networks, e.g. restoration can vary from simple methods (e.g. managing weeds) to more costly and time-demanding actions (e.g. creating new ecosystems) (Wintle et al., 2011). (v) Targeted gene flow, where individuals of the same species that are pre-adapted to future conditions (e.g. species that are more exposure to hotter dried climates) are translocated to increase the adaptive capacity of another population (Macdonald et al., 2017). Despite of the multiple innovative approaches to improve biodiversity conservation strategies (Reside et al., 2018), the first step to improve the effectiveness of natural protected areas and biodiversity conservation in managed landscapes, is to identify new priority conservation areas considering multiple taxonomic groups and different spatial levels. Our study showed that it is possible to take advantage of the different current biodiversity studies at landscape level, and combine this information to create spatial decision-support maps to assist the existing biodiversity conservation strategies. Moreover, it is possible to combine these maps with others studies (e.g. human footprint maps) to achieve different goals (e.g. potential areas with high wilderness values) (Rosas et al., 2021b).

## 5. Conclusions

The first step in conservation planning framework is to identify conservation priority areas. ENFA indexes (marginality and specialization) and potential habitat suitability maps obtained with the Biomapper software allow us to understand ecological requirement of taxonomic species groups, and to build two final spatial decision-support maps for conservation strategies. Maps of potential biodiversity (MPB) provide taxonomic group distributions related to environmental conditions, and identified hotspot areas of relevant species. The maps of priority conservation areas (MPCA) allow us to define conservation priority areas by considering endemism and richness at different scale levels. Taxonomic groups presented different ecological and environmental requirements: (i) lizards and darkling-beetles were mostly related to dry steppes and temperature variables, (ii) birds and plants were related to different vegetation types (e.g. humid steppes, shrublands, *Nothofagus antarctica* forests, and ecotone areas) associated with high values of NDVI and different temperature regimes, and (iii) the studied mammal was mainly associated to *N. pumilio* forests and alpine vegetation. At provincial level, both spatial decision-support maps highlight the importance of overlapping different species requirements in shrublands and humid steppes, while MPCA identify high values related to forests and alpine vegetation due to endemism (e.g. *Hippocamelus bisulcus*). However, only MPB highlight the importance of *N. antarctica* forests for conservation inside the forest landscape matrix. In addition, both methodologies expose the effectivity of current natural protected networking, where MPB presented the highest values in unprotected areas, but MPCA highlight the importance of the protected areas. We considered that steppes and shrublands areas, as well as, N. antarctica forests need to be prioritized for new protected areas and to increase sustainable economic strategies outside of the current protected areas (land-sharing strategy). However, we think that before to recommend new specific protected areas, it is necessary to complement this study with other information (e.g. economic, social and political studies) and to reach an agreements with other sectors and stake holders (policy makers, farmers, watershed institutions, etc) that are outside of the scope of this study. Spatial decision-support maps can contribute to improve biodiversity conservation strategies considering different scale levels, including areas with low coverage of data-bases as Patagonia.

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## Authors contributions

Yamina M. Rosas: Conceptualization, Methodology, Investigation, Writing – original draft. Pablo L. Peri: Data curation, Writing – review & editing. María V. Lencinas: Data curation, Methodology, Formal analysis, Writing – review & editing. Leónidas Lizarraga: Writing – review & editing. Guillermo J. Martínez Pastur: Investigation, Formal analysis, Methodology, Data curation, Writing – review & editing.

## Declaration of competing interest

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

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2022.114578.

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