Chapter 2
Assessment of Provisioning Ecosystem Services in Terrestrial Ecosystems of Santa Cruz Province, Argentina

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Abstract Provisioning ecosystem services play an important role in the development of regional economies. Traditional managements usually intensify the supply of provisioning services, without consideration of other services (e.g. cultural and supporting) and biodiversity. The objective of this chapter was to characterize main provisioning ecosystem services and potential biodiversity in different terrestrial ecosystems (native forests, shrublands and grasslands) of Santa Cruz Province (Southern Patagonia, Argentina) and to identify potential trade-off areas between provisioning ecosystem services and biodiversity conservation values. We found that non-forested areas exhibited higher supply of provisioning ecosystem services and biodiversity values than forested areas, where potential trade-off areas were located in humid steppes and shrublands. Particularly, in *Nothofagus* forests landscape, provisioning ecosystem services and biodiversity increased with forest cover, where *N. antarctica* forests type showed more potential trade-off areas than other *Nothofagus* forests type, while new potential protected areas were located when different forest types were combined (*N. antarctica* and *N. pumilio*). These results can be used by decision-makers to improve management and conservation strategies on private lands.

Keywords Ecosystem services · Biodiversity values · Ecological areas · *Nothofagus* forest types · Trade-offs · Landscape scale

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1 Introduction

Natural ecosystems provide multiple services and goods to people, usually named as ecosystem services (ES) (MEA 2005). There are different ES (provisioning, supporting, regulating and cultural) and are known as provisioning ecosystem services (PrES), the most important ES for societies (Ala-Hulkko et al. 2019). Natural capital includes all natural resources that society uses. ES are provide by biotic organisms or an interaction with abiotic processes (Haines-Young and Potschin 2018). In this context, the last version of ES classification (CICES v.5) includes abiotic ES related to mineral substances that are used as energy sources (e.g. crude fossil fuels). The cascade model for the landscape in Southern Patagonia proposed by Rosas et al. (2019a) linked the forest ecosystem with social systems and identified the potential synergies (positives and negatives) between management and conservation planning. The different terrestial ecosystems of the region determined specific PrES related to biophysical characteristics (e.g. climate, topography and vegetation), and where policy decisions impacted on how these services were obtained through the management strategies implementation (Peri et al. 2016a, b, c; Perera et al. 2018).

Patagonian ecosystems (e.g. steppes and native forests) provide different ES to local people (Laterra et al. 2011). However, when ecosystems are managed only to maximize PrES, many other ES (e.g. regulating or cultural) and biodiversity values are usually undervalued (Thompson et al. 2011; Oñatibia et al. 2015; Martínez Pastur et al. 2017; Perera et al. 2018). Sheep production is one of the most important economic activities in Santa Cruz Province based on extensive grazing and provide PrES as lamb and wool animal. Different studies analysed sheep breeding in the province and determined sheep-carrying capacity (Andrade et al. 2016), trade-off between livestock and biodiversity (Pedrana et al. 2010; Peri et al. 2013; Travaini et al. 2015) and the impact of grazing on soil properties (Peri et al. 2016a). Oil production is also another important economic activity and provides abiotic PrES as crude oil, where the wells establishments, accessibility infrastructure, pipelines and other oil facilities presented different impacts on natural areas by generating habitat fragmentation (Buzzi et al. 2019) and increasing potential desertification processes (Del Valle et al. 1998; Gaitán et al. 2019). Fiori and Zalba (2003) determined that vegetation recovery on seismic lines is extremely poor and facilitates expansion of exotic invasive plants.

In addition, native forest ecosystems provide PrES such as timber wood, fibre or firewood (Gea et al. 2004), food (e.g. fruits, nuts, mushrooms, honey or spices), pharmaceutical plants and other non-woody industrial products (MEA 2005). Studies had been developed to determine timber production of different *Nothofagus* species, especially for *N. pumilio* forests (NP) (Peri et al. 2019a), and to define new silvicultural proposals (Gea et al. 2004; Martínez Pastur et al. 2009, 2019) considering different economics and conservation values. In addition, silvopastoral systems, which combine trees and grasslands or pastures under grazing in the same unit of land, became an economical, ecological and social productive alternative in N. antarctica forests (NA) (Peri and Ormaechea 2013; Peri et al. 2016b), which combine trees and grasslands or pastures under grazing in the same unit of land, became an economical, ecological and social productive alternative in Patagonia. Silvopastoral systems are designed to increase the provision of ES from managed forests, such as livestock (e.g. cattle, goats and sheep) that generates different products (e.g. meat, milk, wool and leather) (Peri et al. 2016b).

In the last years, the interest to understand the relationship between ES supply and biodiversity had increased (Currie 2011; Mace et al. 2012; Maes et al. 2014). Biodiversity had been defined as critical to support ES delivery (Mori et al. 2017) through its role in functional processes (Thompson et al. 2011; Harrison et al. 2014). In fact, some authors suggested that biodiversity itself can be considered as an ES (Mace et al. 2012). In Santa Cruz Province, there are antecedents related to conservation of emblematic species (e.g. *Lama guanicoe*) (Pedrana et al. 2010; Travaini et al. 2015), endangered species (e.g. *Hippocamelus bisulcus*) (Vila et al. 2006; Flueck and Smith-Flueck 2012) and endemic species of darkling beetles (Carrara and Flores 2013) and lizard (Breitman et al. 2014).

Understanding the connections between ES (especially PrES) and biodiversity has been a challenge due to multiple (e.g. ecological, social and scales) perspectives (De Groot et al. 2010; Thompson et al. 2011), mainly in remote areas due to lack of data (Martínez Pastur et al. 2017). Recent methodologies have improved the assessment of species distributions, synergies and trade-offs among ES and biodiversity at different spatiotemporal scales (Raudsepp-Hearne et al. 2010; Cordingley et al. 2016) using scarce available data from field works and remote sensing approaches (Martínez Pastur et al. 2016b).

In the Patagonian region, some studies analysed the impacts of livestock on plants biodiversity (Peri et al. 2013, 2016a, c) and changes on arthropods richness and abundance (Sola et al. 2016; Lescano et al. 2017). In addition, Rosas et al. (2019a) tried to describe the importance of these connections in Southern Patagonian forests, as well as some studies of plant and insect assemblages in non-managed (Peri and Ormaechea 2013; Peri et al. 2019a, b) and harvested *Nothofagus* forests (Gargaglione et al. 2014). During the last years, several studies conducted in Southern Patagonia reported maps of supporting, regulating (Peri et al. 2018, 2019b) and cultural ES (Martínez Pastur et al. 2016a; Rosas et al. 2019a). In addition, potential biodiversity maps (PBM) combining potential habitat suitability maps of different taxa were developed (Martínez Pastur et al. 2016b; Rosas et al. 2018, 2019b, c). PBM, that synthetize the information of several species, became a useful tool to define better management and conservation planning (Rosas et al. 2019b), define the effectiveness of the current protected areas network (Rosas et al. 2018), and identify hotspot areas (Rosas et al. 2019c) and different trade-offs among ES and biodiversity (Martínez Pastur et al. 2017).

In this context at landscape level, the main challenge is to decide the best option of land use management (production and/or conservation) (Carpenter et al. 2009; Raudsepp-Hearne et al. 2010; Cordingley et al. 2016). Mapping methodologies had been used to support policy decisions (De Groot et al. 2010; Maes et al. 2012) by incorporating landscape heterogeneity (Martínez Pastur et al. 2017). Land-use decisions depend on public policies such as the national law no. 26331/07 that defined forest areas under different uses (timber, restoration, conservation). However, the use of these information (e.g. supply of PrES) by public and private policies is scarce (Braat and De Groot 2012). PrES and biodiversity integration analysis may improve the current conservation plans (e.g. identify areas with the highest PBM values), increase the landscape multi-functionality (e.g. combination of sustainable economic activities) or reduce economic costs of companies (Raudsepp-Hearne et al. 2010; Mori et al. 2017).

The objective of this chapter was to analyse the different provisioning ecosystem services (PrES) and potential biodiversity (PBM) in terrestrial ecosystems of Santa Cruz Province (Southern Patagonia, Argentina), with special emphasis on *Nothofagus* forest landscapes. Also, we aimed to identify the (i) potential trade-offs between PrES and MPB outside of the networking protected area, (ii) areas with high MPB and low PrES values to suggest new potential protected areas and (iii) areas with low MPB and high PrES values where conflicts are low and intensification of the management activities is possible.

2 Study Case in Santa Cruz Province

2.1 Study Area

Southern Patagonia includes Santa Cruz Province (Argentina), which covers 243,943 km^2 (Fig. 2.1a) and presents a variety of terrestrial ecosystems dominated by dry steppes in the north and centre; humid steppes and shrublands in the south; and sub-Andean grasslands, *Nothofagus* forests and alpine vegetation occupying a narrow strip near the Andes mountains (Oliva et al. 2004) (Fig. 2.1b). The province presents 7% of the total area under protection (Fasioli and Díaz 2011), while most of the areas are private lands (93%). National parks mainly preserve forests and ice fields close to the mountains in the west (e.g. Perito Moreno National Park), and provincial reserves mainly protect special features in the steppe landscape (e.g. Meseta Espinosa y El Cordón Provincial Reserve) (Fig. 2.1c). Despite this, most of the protected area networks are located near the Andean mountains, where *Nothofagus* forest types are not equally protected (Rosas et al. 2019a). These forests types are distributed from 46° to 52° SL, in a wide range of rainfall, temperature patterns and elevation gradients (Veblen et al. 1996; Peri and Ormaechea 2013; Peri et al. 2019a) (Fig. 2.1d). Detail of *Nothofagus* forests, which names are related to lakes and cities, showed *Nothofagus* forest types distribution (Fig. 2.1d I, II, III, IV and V), where 69% of NP forests (2246 km^2) are protected and mainly distributed in the north and central areas of the province, 82% of mixed evergreen forests (180 km^2) are protected in central areas. NA forests (1699 km^2) prevail in the southern area and only 16% are under protection.

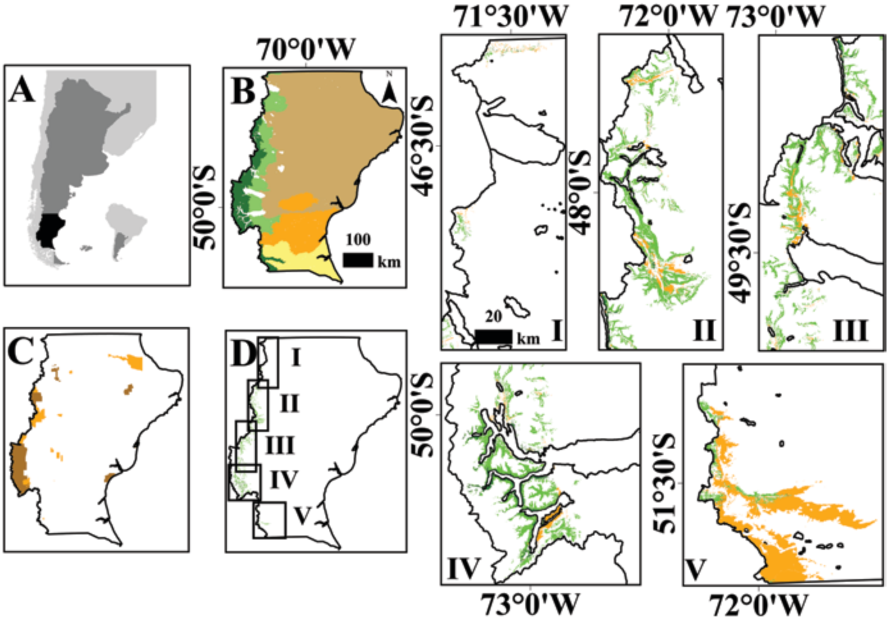

Fig. 2.1 Characterization of the study area: (**a**) location of Argentina (dark grey) and Santa Cruz Province (black); (**b**) main ecological areas (brown = dry steppe, yellow = humid steppe, orange = shrublands, light green = sub-Andean grasslands, dark green = *Nothofagus* forests and alpine vegetation) (modified from Oliva et al. 2004); (**c**) protection areas (orange = provincial reserves, brown = national parks); (**d**) *Nothofagus* forests (light green = *N. pumilio*, orange = *N. antarctica*, dark green = mixed forests) (CIEFAP-MAyDS 2016); more detail of *Nothofagus* forests: (I) Lago Buenos Aires, (II) Lago Pueyrredón, (III) Lago San Martín, (IV) Lago Argentino, (V) Río Turbio

Table 2.1 Proxies and units of provisioning ecosystem services

Type	Division	Proxy	Unit
Provision	Nutrition	Sheep presence probability	Probability of sheep presence km^{-2}
		Potential silvopastoral	Index
	Plants and fibre	Total volume without bark	$m^3\ ha^{-1}$
	Oil production	Oil well density	Well km^{-2}

2.2 *Materials and Methods*

2.2.1 Provisioning Ecosystem Services Map

We elaborated one provisioning ecosystem services map (PrESM) considering four proxies based on CITES divisions (MEA 2005; Maes et al. 2012, 2014; Haines-Young and Potschin 2018) (Table 2.1).

Each proxy map was built using a geographical information system (GIS) project and rasterized at 90 × 90 m resolution using the nearest resampling technique:

(i) Sheep probability map was calculated using sheep stocking density estimated from a model of probability of contact with sheep per ranch (0–1 probability km^{-2}) according to Pedrana et al. (2011), where values close to 0 indicate low probability of occurrence and values close to 1 indicate the highest probability of occurrence. In the GIS project, we applied the focal statistics tool to create a new raster by considering the values near 10 km, and then we applied a mask where forests and protected areas had values of zero sheep probability.
(ii) Oil production map was estimated based on oil well density (wells km^{-2}). In the GIS project, we calculated the oil well density using 21,426 points of well (http://datos.minem.gob.ar//). We did not apply a mask with zero values inside protected areas, because oil activities is legal (Law n° 2.185), we did not apply a mask with zero values.
(iii) Timber production map was calculated as potential total volume without bark (TVWB m^3 ha^{-1}) of NP and mixed evergreen forests according to the provincial forest inventory (Peri et al. 2019a). NA forests, non-forests and protected areas presented value of zero timber production.
(iv) Potential silvopastoral map was calculated using understory biomass production (kg DM ha^{-1}) and total volume without bark (m^3 ha^{-1}) of NA forests (Peri and Ormaechea 2013). These authors defined that understory biomass production varied from <500 to >2500 kg DM ha^{-1} and total volume without bark varied from <100 to >200 m^3 ha^{-1}. In the GIS project, we applied the reclassify tool to classify the rasters from 1 to 4. Then, we calculated a potential silvopastoral index considering that 70% of NA forests had livestock and 30% is used to obtain poles wood or firewood (potential silvopastoral index = biomass production × 0.7 + total volume without bark × 0.3). Then, the equation was integrated into the GIS project. We applied a mask where NP and mixed evergreen forests, non-forests and protected areas represent zero potential silvopastoral value.

The four proxy maps were rescaled from 0 to 100 and combined (sum values for each pixel) to obtain the final PrESM. This map was rasterized to present scores that varied from 0 to 100.

2.2.2 Potential Biodiversity Map

We elaborated a potential biodiversity map (PBM), using 119 potential habitat suitability maps of different taxonomic group species (Rosas et al. 2017, 2018, 2019b, c). These maps used a large database: (i) one endangered mammal (*Hippocamelus bisulcus*) in *Nothofagus* forests by using 300 plots from National Park Administration and different studies (Vila et al. 2006); (ii) 47 species of birds by using 5512 plots (Darrieu et al. 2009) and one international web platform of bird collection (https://ebird.org/); (iii) 7 species of lizards by using 250 plots (Cruz et al. 2005; Ibargüengoytía et al. 2010; Fernández et al. 2011; Breitman et al. 2014); (iv) 10 species of darkling beetles by using 310 plots from CEI (Colección Entomológica del Instituto Argentino de Investigaciones de las Zonas Áridas, IADIZA) and (V) 53 species of vascular plants by using 5915 plots from PEBANPA Network (Peri et al. 2016c), native forests provincial inventories and data from FAMA INTA laboratory (Forestal, Agricultura y Manejo del Agua). The database also was complemented with data of the selected species using the Sistema Nacional de Datos Biológicos of Ministerio de Ciencia, Tecnología e Innovación Productiva (www.datosbiologicos.mincyt.gob.ar). Environmental Niche Factor Analysis (ENFA, Hirzel et al. 2002) and Biomapper 4.0 software (Hirzel et al. 2004) were used for species potential habitat suitability mapping based on 41 potential explanatory variables (climate, topography, and other variables related to landscape), which were rasterized at 90 × 90 m resolution using the nearest resampling technique on a GIS project. The GIS methods used here were described in Rosas et al. (2017, 2018, 2019b, c). The maps for each taxonomic group species were combined (average values for each pixel) to obtain four potential biodiversity maps (birds, lizards, darkling beetles and plants) and one potential habitat suitability (mammal). We used a mask of NDVI <0.005 to exclude ice, water or bare soil. The five maps were weighted by a group importance index from 0.5 to 1.0 that combined ENFA index (Hirzel et al. 2002) and endemism of each species. The five weight maps were rescaled from 0 to 100 and then combined (sum values for each pixel) to obtain the final PBM for the province. This map also was rasterized to present scores that varied from 0 to 100.

2.2.3 Landscape Analyses

We calculated the mean of each PrES proxies, PrESM and PBM using a hexagonal binning processes (each hexagon = 250,000 ha) for the full province and for forest landscape matrix (each hexagon = 5000 ha). We analysed the maps considering the influence of the different ecological areas (Oliva et al. 2004) and forest landscape matrix (combination of grasslands and the different forest types) (Peri and Ormaechea 2013; Peri et al. 2019a) by using one-way ANOVAs and Tukey post-hoc test. The hexagonal GIS methods used here were previously described by Rosas et al. (2019c).

Additionally, we analysed the performance of PrESM to detect potential trade-offs with the biodiversity for the total area, forest landscapes and main forest types. For this, we categorized the PBM (low, medium and high) considering equal number of hexagons. For the whole province, the thresholds of potential biodiversity were as follows: low <41%, medium 42–74% and high <75%, and for forest landscape, the thresholds were as follows: (i) G – low <35%, medium 36–47% and high <48%; (ii) G + F – low <52%, medium 53–62% and high <62%; (iii) F – low <67%, medium 68–76% and high <77%. Also, based only on G + F and F hexagons, we classified each one considered the main forest types (NA and NP), according to the most abundant forest type cover inside each one. Finally, we want to identify the (i) potential trade-offs outside the networking protected areas, (ii) potential new protected areas (high MPB and low PrES values) and (iii) potential areas where the economic activities are maximized through intensive management (low MPB and high PrES values). For this, we built a new map crossing PBM (low, medium and high) and PrESM (low and high) categories. These new maps were classified considering equal number of hexagons: (i) for the entire province, the selected thresholds were as follows: PBM – low <41%, medium 42–74% and high <75%; PrESM – low <24% and high >25%; (ii) for forest landscape matrix, the selected thresholds were as follows: PBM – low <41%, medium 42–54% and high <55%; PrESM – low <22% and high >23%.

2.3 Results and Discussion

2.3.1 Provisioning Ecosystem Services Map

Sheep presence probability and oil well density proxies in Santa Cruz province occurred in most of the ecological areas (Fig. 2.2), while total volume without bark and potential silvopastoral proxies were specifically from *Nothofagus* forests (Fig. 2.3).

Sheep presence probability map presented values from zero (e.g. natural protected areas) to 1.00 (e.g. best grazing areas) (Fig. 2.2a). The provincial mean value was 0.41 sheep presence probability km^{-2}, where 19% of the area had low values (<0.20), 60% showed values between 0.20 and 0.80 and 14% presented high values (<0.80). Sheep probability values decreased from south to centre where steppes prevailed and from east to west where sub-Andean grasslands dominated (Oliva et al. 2004). ANOVAs showed that sheep probability map presented significant differences among the different ecological areas (Table 2.2), where humid steppe and shrublands had the highest values (0.71 and 0.68, respectively), followed by the dry

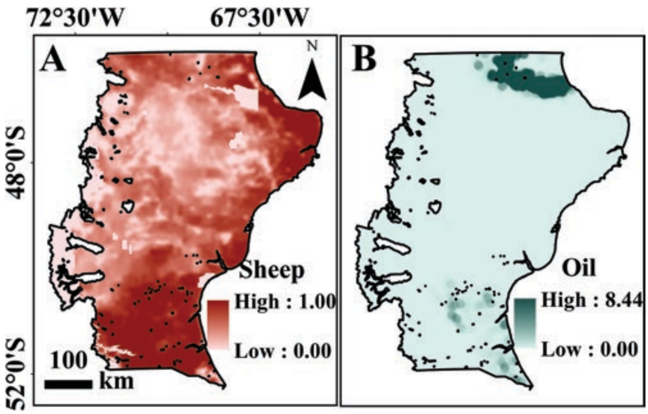

Fig. 2.2 Provisioning ecosystem services of Santa CruzPprovince: (**a**) sheep probability (probability of sheep presence km^{-2}), where dark red represents the highest values and light red the lowest probabilities values and (**b**) oil production (wells km^{-2}), where dark blue represents the highest density values and light blue the lowest density values

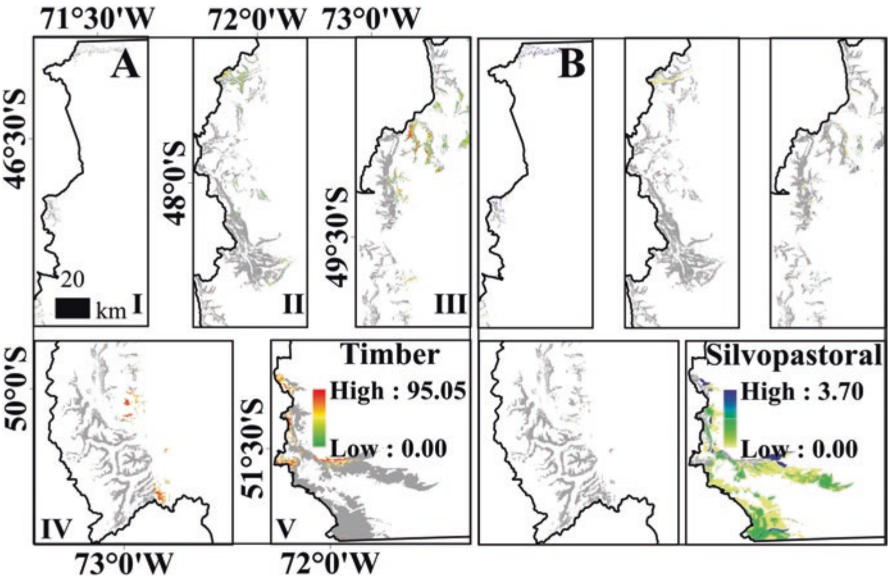

Fig. 2.3 Provisioning ecosystem services from *Nothofagus* forests of Santa Cruz Province: (**a**) timber production of *N. pumilio* and evergreen mixed forest (TVWBm^3 ha^{-1}), red represents the highest volume values and green the lowest volume values and (**b**) potential silvopastoral of *N. antarctica* forest (adimensional), blue represents the highest values and yellow the lowest values. Details of *Nothofagus* forests: (I) Lago Buenos Aires, (II) Lago Pueyrredón, (III) Lago San Martín, (IV) Lago Argentino, (V) Río Turbio. Grey colour indicates *Nothofagus* forests with 0 value

steppe (0.38), and the lowest values were found in sub-Andean grasslands, and *Nothofagus* forests and alpine vegetation (0.21 and 0.07, respectively).

Several studies evaluated the impact of livestock grazing on ecosystem properties (e.g. soil variables and vegetation cover; Peri et al. 2016a) in rangelands, where their extension and economic importance highlight the necessity of sustainable management proposals to supply the demand of an increasing human population (Ala-Hulkko et al. 2019). In Patagonia, extreme climatic condition was used as the most important predictor together with the land-use management in modelling soil carbon concentrations (Peri et al. 2016a). Also, Peri et al. (2013) reported that grass vegetation cover decreased and soil erosion increased due to high livestock stocking rates under continuous grazing in the studied area. In the last 70 years, the degradation of Patagonia steppe had increased due to an inadequate land management (e.g. overgrazing, heterogeneous and large paddocks and continuous grazing) (Del Valle et al. 1998; Gaitán et al. 2019). In addition, sheep presence probability presented significant differences among forest landscape matrix, where the highest values (0.17) were found in the grassland areas (grassland cover >70%). Despite this, when grasslands with forest matrix (G + F) were considered, the combination with NA forests presented the highest values (0.38) as well as when only forest cover (F) was considered. Ecotone areas of NA forests with grasslands have been identified as very important zone for livestock production, where forage species and tree cover

Table 2.2 ANOVAs of different provisioning ecosystem services for terrestrial ecosystems of Santa Cruz Province, considering different ecological areas and forest landscape matrix (grasslands and forests, grasslands and forests types and forest types)

Terrestrial ecosystems			Sheep probability (sheep. km^{-2})	Oil production (wells. km^{-2})	Timber production (TVWB m^3.ha^{-1})	Potential silvopastoral (adimensional)
Ecological areas	Forests and alpine vegetation		0.07 a	0.00	0.60 b	0.08 b
	Humid steppe		0.71 c	0.03	0.21 a	0.04 ab
	Dry steppe		0.38 b	0.12	0.00 a	0.00 a
	Shrublands		0.68 c	0.03	0.05 a	0.00 a
	Sub-Andean grasslands		0.21 a	0.00	0.04 a	0.00 a
	F(p)		28.58 (<0.001)	0.53 (0.716)	9.96 (<0.001)	4.41 (0.002)
Forest landscape matrix	(i) Grasslands and forests	G	0.17 b	0.00	0.38 a	0.02 a
		G + F	0.10 a	0.00	2.29 b	0.09 b
		F	0.06 a	0.00	1.39 b	0.41 c
		F(p)	8.55 (<0.001)	1.73 (0.178)	18.55 (<0.001)	95.99 (<0.001)
	(ii) Grasslands and forest types	G + NP-MIX	0.01 a	–	1.19	0.00 a
		G + NP	0.04 ab	–	2.45	0.00 a
		G + NA-NP	0.12 b	–	3.74	0.12 b
		G + NA	0.38 c	–	0.43	0.49 c
		F(p)	26.83 (<0.001)	–	1.44 (0.237)	67.78 (<0.001)
	(iii) Forest types	NP-MIX	0.00 a	–	0.00	0.00 a
		NP	0.02 a	–	1.57	0.01 a
		NA-NP	0.07 ab	–	3.29	0.14 a
		NA	0.10 b	–	0.38	0.84 b
		F(p)	3.41 (0.023)	–	2.50 (0.068)	63.54 (<0.001)

G grasslands, *F* forests, *NA Nothofagus antarctica*, *NP N. pumilio*, *MIX* mixed evergreen forests
F Fisher test, *(p)* probability. Different letters show differences with Tukey test at $p < 0.05$

increased the habitat qualities for animals (e.g. nutrition properties and shelter for animals) (Peri et al. 2013).

Oil well density map presented values from 0 (minimum density) to 8.44 (maximum density) (Fig. 2.2b), with a mean provincial value of 0.09 wells km^{-2}. The highest values occurred mainly in two areas of the province, one near San Jorge Gulf in the northeast and the other area in the southeast area of the province near Rio Gallegos city, where Producción Petrolera Nacional del Petróleo (ENAP) and Yacimientos Petrolíferos Fiscales (YPF) y TOTAL S.A. are the principal operators. ANOVAs showed that there were not significant differences among different ecological areas ($F = 0.53$; $p = 0.716$) or across the forest landscape matrix. Despite the non-significant differences among ecological areas, there was an increase of oil production values in the dry steppes, where punctual activities (e.g. wells, accessibility and pipelines) presented highest impacts (Del Valle et al. 1998; Gaitán et al. 2019) in protected areas (e.g. Meseta Espinosa y El Cordón Provincial Reserve). This indicated potential trade-off with the conservation of endemic species, where Fiori and Zalba (2003) determined that vegetation recovery in pipelines and oil well areas was extremely poor, being oil the only provisioning ES enable to be conducted inside the protected areas. This creates a potential trade-off with the conservation of endemic species.

Total volume without bark (Fig. 2.3a) varied from 0 (NA forest, open lands and protected areas) to 95.05 (maximum volume without bark), with a mean provincial value of 0.06 TVWB m^3 ha^{-1}. NP and mixed evergreen forests (459 km^2) presented values from 0.01 to 95.05 TVWB $m^3.ha^{-1}$. In Santa Cruz, while 52% of native forests presented low values (<30 TVWB m^3 ha^{-1}), 43% had values between 30 and 60 TVWB m^3 ha^{-1} and only 5% of these forests presented high values (>60 TVWB m^3 ha^{-1}) (Fig. 2.3a). Timber production values increased from north to south. In the north, at the Lago Buenos Aires area (Fig. 2.3aI), all native forests are inside natural reserves, and in the centre areas at Río Chico and Lago San Martin, forests presented values from low to medium (Fig. 2.3aII and III). In the south, Lago Argentino and Río Turbio (Fig. 2.3aIV and V) showed values from medium to high, where the highest values were presented near ecotone areas with the humid steppes. As it was expected, ANOVAs showed significant differences in timber production among ecological areas (Table 2.2), where *Nothofagus* forests presented the highest values (0.60 TVWB m^3 ha^{-1}).

Forest landscape matrix analysis showed that timber production was significantly highest when grasslands were combined with forests (2.29 TVWB m^3 ha^{-1}) or where only forest occurred (1.39 TVWB m^3 ha^{-1}). Furthermore, there was no significant differences among forest types. The use of native forests for timber occurred in the Patagonian region since the European colonization in the late nineteenth century, where harvesting for sawmills and firewood still continues in Tierra del Fuego (Gea et al. 2004; Martínez Pastur et al. 2019). In Santa Cruz Province, there are not operating sawmills (Peri et al. 2019a), and most of the NP (69%) and mixed evergreen (82%) forests are inside the protected areas, where other ecosystem services (e.g. cultural) prevail and mostly define the use of the natural forests (Rosas et al. 2019a).

Potential silvopastoral map (Fig. 2.3b) presents values from 0.00 (e.g. NP and mixed evergreen forests, non-forest and protected areas) to 3.70 (e.g. maximum potential silvopastoral in NA forests), with a provincial mean value of only 0.01. NA forests (1432 km^2) presented values from 1.00 to 3.70, where 93% of the area presented low values (<1.60), 6% had values between 1.70 and 2.30 and only 1% showed high values (>2.40) (Fig. 2.3b). Potential silvopastoral values increased from north to south, where Lago Buenos Aires (Fig. 2.3bI) and Lago Argentino (Fig. 2.3bIV) had the lowest values. Lago Pueyrredón and Lago San Martin (Fig. 2.3bII and III) had medium values at low hillside near valleys and lakes. In Río Turbio (Fig. 2.3bV), the proxy presented the highest values near ecotone areas and lowest values in the extreme south of the province.

ANOVAs showed that potential silvopastoral presented significant differences among the different ecological areas (Table 2.2), where *Nothofagus* forests had the highest values (0.08) followed by humid steppes (0.04). In addition, potential silvopastoral presented significant differences among the forest landscape matrix, where the highest values (0.84) were found in NA forests (forest cover >50%). This is because silvopastoral systems combine trees and grasslands or pastures under grazing in the same unit of land, being an economical, ecological and social productive alternative in Patagonia (Peri et al. 2016b). Peri and Ormaechea (2013) identified that more than 90% of NA forests presented silvopastoral activities. This system provides increasing incomes to ranchers due to the combined production of timber and animals and benefits such as the provision of livestock shelter, enhancement of animal welfare and other beneficial effects on soil conservation (Peri et al. 2016b).

2.3.2 Provisioning Ecosystem Services and Potential Biodiversity Map: Identification of Conservation Areas of Interest

The rescale (0–100) of the four proxy maps allowed us to combine them (sum values for each pixel) and develop the final PrESM (Fig. 2.4a), which presented values from 0 (minimum provisioning ecosystem services) to 100 (maximum provisioning ecosystem services) across the landscape. PrESM increased from north to south and decreased from east to west. Medium to high values occurred near seacoast and humid steppe areas, while the lowest values were located near glaciers and mountain areas. In addition, we combined the 119 potential habitat suitability maps of the different taxonomic group species (Rosas et al. 2017, 2018, 2019b, c) to develop the final PBM (Fig. 2.4b), where values varied from 0 (minimum potential biodiversity) to 100 (maximum potential biodiversity). In general, PBM presented similar pattern as PrESM, with medium to high values obtained from the seacoast to the centre of the province.

ANOVAs showed significant changes in PrESM and PBM across different ecological areas (Table 2.3), where the highest values were found at humid steppes (51.87 and 63.77, respectively) and shrublands (43.01 and 66.66, respectively), while the lowest values in Sub-Andean grasslands (12.20 and 35.32, respectively).

In particular, humid steppes and shrublands showed the sheep breeding proxy as the most important PrES, and PBM presented different plant species that highlighted the importance of these areas (e.g. *Carex* spp. and *Festuca pallescens*). Different studies have been developed to understand the plant biodiversity distribution and their importance on the ecosystem function (Peri et al. 2013; Gaitán et al. 2014) and economic activities (Peri et al. 2013). However, few studies focused on biodiversity related to grazing in these ecosystems (Peri et al. 2016c). Nevertheless, potential trade-offs between forage provision and regulating and supporting services (e.g. carbon and nitrogen stocks) have been observed (Oñatibia et al. 2015; Peri et al. 2016a). In fact, negative consequences (e.g. desertification) (Del Valle et al. 1998; Gaitán et al. 2019; Peri et al. 2016a) due to overgrazing (Peri et al.

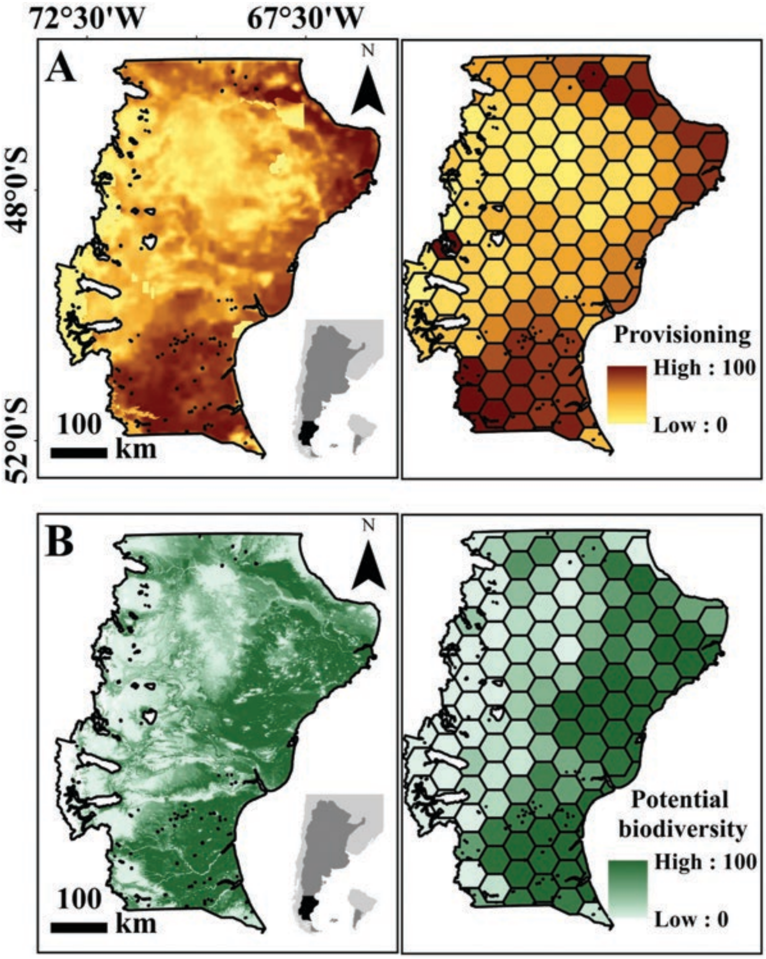

Fig. 2.4 Provisioning ecosystem services and potential biodiversity maps (0–100) of Santa Cruz Province (left) and hexagons of 250,000 ha obtained through the hexagonal binning process (right). (**a**) Provisioning ecosystem services, brown represents the highest values (values close to 100) and yellow the lowest values (values close to 0) and (**b**) potential biodiversity map, dark green represents greater potential biodiversity and light green the lowest potential

2016c), land-use conversion and climate changes (Gaitán et al. 2019) have been reported for the steppe ecosystem. In this context, endemic species with a narrow distribution and high potential habitat suitability values (e.g. *Nyctelia bremi, Liolaemus sarmientoi*) with very specific environmental conditions became an important issue for conservation (Rosas et al. 2018; Rosas et al. 2019b) mainly in areas with a lack of protected areas (e.g. humid steppes). Additionally, bird species presented conflicts with specific economic practices, e.g. shrub removal (e.g. *Junielia tridens*) to increase grasses biomass, which may affect reproductive

Table 2.3 ANOVAs of provisioning ecosystem services (PrESM) and potential biodiversity (PBM) maps in terrestrial ecosystems of Santa Cruz Province (0–100), considering different ecological areas and the forest landscape matrix (grasslands and forests, grasslands and forests types and forest types)

Terrestrial ecosystems			PrESM	PBM
Ecological areas	Forests and alpine vegetation		23.70 a	43.61 a
	Humid steppe		51.87 b	63.77 bc
	Dry steppe		24.49 a	57.39 b
	Shrublands		43.01 b	66.66 c
	Sub-Andean grasslands		12.20 a	35.32 a
	F(p)		12.82 (<0.001)	17.95 (<0.001)
Forest landscape matrix	(i) Grasslands and forests	G	13.74 a	39.70 a
		G + F	17.07 a	48.19 b
		F	28.38 b	54.89 c
		F(p)	12.10 (<0.001)	174.46(<0.001)
	(ii) Grasslands and forest types	G + NP-MIX	4.31 a	45.22 a
		G + NP	10.02 a	46.00 a
		G + NA-NP	24.15 a	52.62 b
		G + NA	46.73 b	49.95 ab
		F(p)	10.41 (<0.001)	9.96(<0.001)
	(iii) Forest types	NP-MIX	0.00 a	46.06 ab
		NP	6.41 a	49.02 a
		NA-NP	20.95 a	54.09 b
		NA	49.05 b	59.90 c
		F(p)	16.68 (<0.001)	17.65 (<0.001)

G grasslands, *F* forests, *NA Nothofagus antarctica*, *NP N. pumilio*, *MIX* mixed forests
F Fisher test, *(p)* probability. Different letters show differences with Tukey test at $p < 0.05$

processes and food for some species (e.g. *Asthenes anthoides, Turdus falcklandii* and *Sturnella loyca*) (Kusch et al. 2016).

Dry steppes presented low values of PrESM (mean of 24.49) and medium values of PBM (mean of 57.39), where both proxies (sheep and oil production) were the most important PrES. These areas occupied more than 60% of the studied province with evident desertification processes (Del Valle et al. 1998) due to the extreme climate conditions and scarce vegetation cover dominated by small shrubs (e.g. *Nassauvia glomerulosa* and *Mulinum spinosum*) and grasses (*Stipa sp*.) (Oliva et al. 2004). Furthermore, oil production greatly affected this area (see Fig. 2.2b) with potential trade-off with the biodiversity. In fact, according to the local regulations, this is the only PrES allowed inside the protected areas (e.g. Mesera Espinosa and El Cordón provincial reserve). These dry steppe areas showed highest values for lizards (e.g. *Liolaemus bribronii*, *L. fitzingerii, Diplolaemus bibronii* and *Homonota darwinii darwinii*) (Breitman et al. 2014; Rosas et al. 2018), where *H. darwinii darwinii* present the most austral gecko distribution. Also, dry steppes presented medium to high values of potential biodiversity for darkling beetles (Rosas et al. 2019b), with high levels of endemism (Carrara and Flores 2013), e.g., for *Nyctelia fitzroyi* which lived in a narrow area with extreme environmental conditions.

The combination of PBM and PrESM allowed us to locate different areas of conservation interest for Santa Cruz Province (Fig. 2.5a). (i) Potential trade-off areas outside the protected areas (high MPB and high PrES values) decreased from east to west, where protected areas were located (Fig. 2.1c, brown to light green colour). We identified high potential trade-off areas near the seacoast (Fig. 2.5a, brown colour), where the biggest area is located in the centre-west (hexagons = 16), followed by one in the south (hexagons = 11) and the smallest area in the north (hexagons = 2) of the province. Another section with medium potential trade-off areas (medium MPB and high PrESM values) was identified across the province (orange colour). (ii) Potential areas to suggest new protected areas (high MPB and low PrES values) were identified in three little sections (dark green colour): one near Monte Leon National Park in the south, another in the steppe areas (hexagons = 3) and the third near Bosques Petrificados at Jaramillo National Park. Medium potential areas to suggest new protected areas (medium MPB and low PrESM values) were identified mainly in dry steppes areas (green colour) and near big lakes (e.g. Lago Argentino). (iii) Potential areas where conflicts are low and

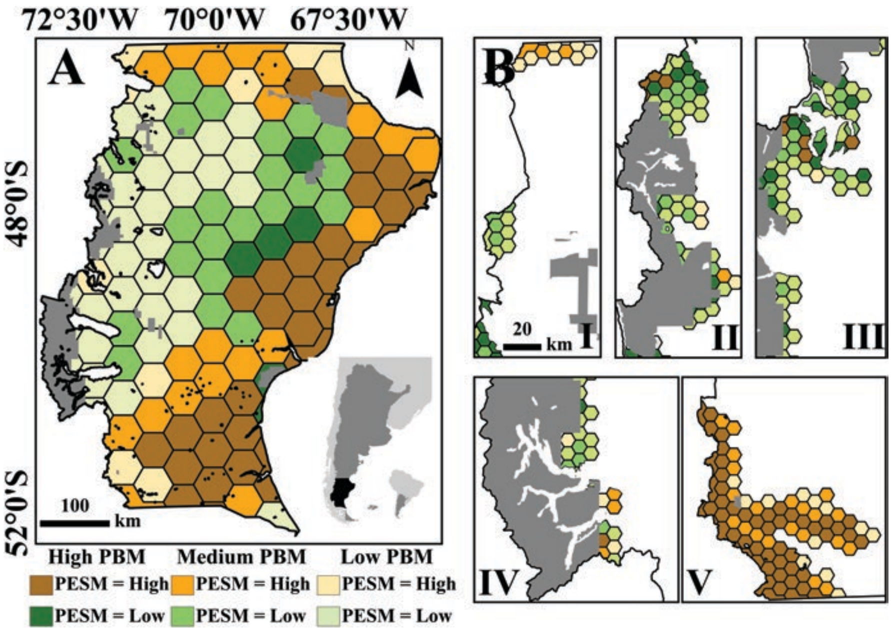

Fig. 2.5 Classification of cross map between PBM (low, medium and high) and PrESM (low and high), considering hexagons of 250,000 ha for Santa Cruz Province (right) and hexagons of 5000 ha for *Nothofagus* forest landscape matrix. Details of *Nothofagus* forests are as follows: (I) Lago Buenos Aires, (II) Lago Pueyrredón, (III) Lago San Martín, (IV) Lago Argentino and (V) Río Turbio, where grey colour indicates protected areas

intensification of the management activities are possible (low MPB and high PrES values) were located in the extreme north and near NA forest in the south (light orange colour) of the province. In fact, ANOVAs showed significant differences of PrESM among the different PBM qualities (low, medium and high) across the province, where PrESM values increased with PBM qualities (Table 2.4).

However, PrESM and PBM showed the lowest values in the west of Santa Cruz Province (see Fig. 2.4 and Table 2.3), where most of the ecosystems are under protection (Fig. 2.1c). Forest landscape matrix ANOVAs showed that the highest values of PrESM and PBM occurred in grasslands with NA forests (forest cover between 30% and 50%) and in areas where NA forests prevail (forest cover >50%) with high sheep probability and potential silvopastoral ESP (Table 2.2). In this context, by crossing PBM and PrESM maps, we located areas of interest for management and conservation planning at landscape level (Fig. 2.5b). (i) Potential trade-off areas decreased from north and south to central-west part of the province (brown to light green colour in Fig. 2.5bV). The biggest area with high potential trade-off (brown colour) is located close to Río Turbio, where NA forests prevail (Fig. 2.1dV). Medium potential trade-offs (orange colour) were identified in Lago Buenos Aires, Lago Argentino and close to Rio Turbio (Fig. 2.5bI, IV, V). (ii) High- and medium-potential areas that suggest new protected areas (dark green and green colour) were located close to Lago Pueyrredón and Lago San Martín (Fig. 2.5bII and III), where few high-potential trade-off areas (brown colour) also occurred due to the existence of different *Nothofagus* forests (NA and NP). (iii) Potential areas where conflicts are low and intensification of the management activities are possible (light orange colour) were located mostly in ecotone areas, where *Nothofagus* forests are combined with grasslands for silvopastoral purposes. In addition, ANOVAs showed that NA forests presented the highest values of PrESM for all PBM qualities (low, medium and high) when the main forest types were considered (Table 2.4).

Forest types characterized by multiple microenvironments allowed the survival of several plant species (Lencinas et al. 2008a; Antos 2009). High potential biodiversity value for understory plants were coincident with studies conducted in Tierra del Fuego (Lencinas et al. 2008a; Martínez Pastur et al. 2016b). In addition, Peri and Ormaechea (2013) identified more shrub and grass species in open than closed NA forests. Our results, identified specific plant species associated with different *Nothofagus* forest types (e.g. *Acaena magellanica, Avenella flexuosa* and *Baccharis magellanica*), and different hotspot areas were identified mainly located in the southernmost part of the province. These hotspot areas presented high-potential habitat suitability values for *Berberis empetrifolia* and *Agrostis capillaris* in NA forests (Rosas et al. 2019c). Silvopastoral management practices increase plant biomass by removing trees and maintaining at the same time more biodiversity values than other proposals (e.g. forest conversion in grasslands) (Peri et al. 2016b). Silvopastoral management also generates positive synergies with biodiversity by enhancing bird, insect and plant richness (Barbier et al. 2008; Peri et al. 2019a). High incoming light levels to understory when canopy trees are removed (thinning practices) provide more energy for plant growth (Antos 2009) and insect richness (Lencinas et al. 2008b). Lantschner and Rusch (2007) also found that birds

Table 2.4 ANOVAs for total provisioning ecosystem services (0–100) and the different ecosystem service types (sheep breeding, oil extraction, timber production and silvopastoral practices) in terrestrial ecosystems of Santa Cruz Province based on different potential biodiversity qualities (low <48%, medium 49–66% and high >100%) for the total province, landscape types (grasslands, grasslands and forests, and forests) and the main forest types

Potential biodiversity			Provisioning	Sheep (sheep.km^{-2})	Oil (wells.km^{-2})	Timber (VTSCm^3.ha^{-1})	Silvopastoral
Total area		Low	18.88 a	26.78 a	1.26	8.03 c	2.03
		Medium	26.63 a	45.86 b	3.24	0.99 a	2.64
		High	38.45 b	69.74 c	5.32	0.00 a	0.00
		F(p)	13.89 (<0.001)	37.35 (<0.001)	0.81 (0.446)	4.31 (0.016)	0.71 (0.495)
Landscapes	G	Low	0.10 a	12.49	–	0.3	0.00 a
		Medium	0.17 b	20.10	–	0.26	0.02 a
		High	0.15 ab	17.20	–	0.58	0.05 b
		F(p)	3.10 (0.051)	2.65 (0.073)	–	2.53(0.081)	15.95 (<0.001)
	G + F	Low	0.06 a	4.52 a	–	0.17 a	0.05 a
		Medium	0.13 a	7.26 ab	–	2.05 ab	0.04 a
		High	0.33 b	16.89 b	–	4.65 b	0.18 b
		F(p)	10.77 (<0.001)	4.31 (0.017)	–	7.41 (0.001)	4.53 (0.014)
	F	Low	0.07 a	3.15	–	0.42	0.09 a
		Medium	0.25 b	7.98	–	1.87	0.32 a
		High	0.54 c	8.42	–	1.96	0.84 b
		F(p)	25.70 (<0.001)	2.11 (0.130)	–	1.27(0.287)	30.48 (<0.001)

(continued)

Table 2.4 (continued)

Potential biodiversity			Provisioning	Sheep (sheep.km^{-2})	Oil (wells.km^{-2})	Timber (VTSCm^3.ha^{-1})	Silvopastoral
Main forest types	NP	Low	0.70 a	0.12 a	–	0.92 a	0.20 a
		Medium	9.69 b	3.55 a	–	12.21 b	1.42 a
		High	40.46 c	14.81 b	–	49.23 c	7.71 b
		F(p)	31.73 (<0.001)	17.20 (<0.001)	–	30.73 (<0.001)	8.23 (0.001)
	NA	Low	39.08	27.21	–	5.36	35.81
		Medium	37.77	19.2	–	7.67	39.58
		High	44.22	13.67	–	6.77	50.95
		F(p)	0.40 (0.673)	1.93 (0.156)	–	0.05 (0.956)	1.12 (0.333)

G grasslands, *F* forests, *NA Nothofagus antarctica*, *NP N. pumilio*
F = Fisher test, *(p)* probability. Different letters show differences with Tukey test at $p < 0.05$

associated of ecotone environments (e.g. forests and grasses areas) moved to managed NA forests and increased the original richness and diversity.

Low values of PrESM (between 24.15 and 20.95) and medium values of PBM (between 52.62 and 54.09) occurred in grasslands with NA-NP (forest cover between 30% and 50%) and NA-NP forest areas (forest cover >50%), where timber production proxy presented the main provision ecosystem service (Table 2.3). There were significant differences of PrESM among different PBM qualities (low, medium and high) when main forest types were considered (Table 2.4). In Santa Cruz Province, plant biodiversity changed through NP forest landscapes (Rosas et al. 2019c). This result was coincident with other studies of NP forests in Tierra del Fuego (Martínez Pastur et al. 2016b); thus values of plant biodiversity increased when different *Nothofagus* forest types were combined (Lencinas et al. 2008a) and insect biodiversity increased in areas with high timber potential (Lencinas et al. 2008b). However, some silvicultural practices (e.g. shelterwood cuts) had negative impacts on insect populations (Spagarino et al. 2001) and increased native and exotic plant species in the harvested areas (Martínez Pastur et al. 2002).

In addition, some mammals such as *Hippocamelus bisulcus* (huemul) can be affected by PrES in NA and NP forested areas (Rosas et al. 2017). Several studies relate the decrease of the huemul habitat to different human impacts (Corti et al. 2013; Briceño et al. 2013) that greatly affected the marginal populations of huemul (e.g. extreme distribution areas in the south and north of the province where silvopastoral activities predominate). Huemul is one of the most vulnerable species with only 350–500 individuals in 50 fragmented subpopulations throughout Patagonia (Díaz and Smith-Flueck 2000), living mainly inside protected areas that represent about 50% of their natural habitat (Vila et al. 2006; Quevedo et al. 2017; Rosas et al. 2017). Therefore, it is necessary to develop new strategies to protect biodiversity outside the protected areas (Mori et al. 2017), where the potential habitat of the huemul is higher (Rosas et al. 2017). Some private initiatives (Ea. Río Condor and Ea. Los Huemules, close to El Chaltén) support this strategy by modifying economic activities inside the ranches, e.g. reducing livestock activity and increasing other activities related to ecotourism. New provincial conservation planning is needed to promote innovative management strategies in productive areas with high-potential habitat suitability values (Smith-Flueck et al. 2011), for example the establishment of corridors and fences to separate cattle and huemul wild populations (Gilbert-Norton et al. 2010; Corti et al. 2011).

The richness of bird species in *Nothofagus* forests were lower in these austral latitudes than in northern hemisphere (Brown et al. 2007; Lencinas et al. 2005); however, most of the species are endemic (e.g. *Agriornis lividus*, *Aphrastura spinicauda* or *Scytalopus magellanicus*). The conservation of forest bird species represented an important challenge for managers, because it is necessary to consider multiple factors such as vegetation structure, connectivity of forest patches with appropriate size and shape to maintain avian diversity, occupancy and turnover rate (Whytock et al. 2018). For this, alternative silvicultural proposals (e.g. variable retention) are necessary to increase the species conservation (Martínez Pastur et al.

2019), where intact patches (e.g. aggregate retention) are combined with single trees in the harvested stands (e.g. dispersed retention). These new proposals maintained some of the original forest structure and micro-environmental conditions in the aggregate patches but increased biodiversity and forest reproduction compared to primary unmanaged forests (Lencinas et al. 2009, 2011; Soler et al. 2016). Variable retention could play a fundamental role for conservation in these forest types, but the influence of retention patterns and the most effective aggregate patch size are still unclear (Martínez Pastur et al. 2019). In this context, the identification of forest areas with potential trade-off between PrES and biodiversity is necessary to develop land-use strategies in the long term (Carpenter et al. 2009; Raudsepp-Hearne et al. 2010; Cordingley et al. 2016).

3 Challenges in the Land-Use Management for Provisioning Ecosystem Services

A key challenge for ecosystem management is to maximize PrES and hold enough biodiversity values across the landscape (Raudsepp-Hearne et al. 2010; Cordingley et al. 2015) to support the society demand (Ala-Hulkko et al. 2019). Some studies showed how human actions improved the delivery of goods (e.g. forage provision) but affected other services (e.g. soil nutrients) or biodiversity (Cardinale 2012; Peri et al. 2016a; Martínez Pastur et al. 2017). According to different studies, it is necessary to protect biodiversity values that are involved in the functional processes and PrES (Thompson et al. 2011; Mace et al. 2012; Maes et al. 2014; Harrison et al. 2014). In the context of spatial and land-use planning, negative and positive interactions had been described between PrES and biodiversity (Cordingley et al. 2016; Turkelboom et al. 2018).

We considered as potential trade-offs those areas with high PrES where intense economic activities affected negatively the provision of another ES and/or the biodiversity conservation. For example, some studies suggested that forest harvesting had potential trade-offs with carbon storage, aesthetic values and habitat quality (Cordingley et al. 2016; Martínez Pastur et al. 2017). In contrast, synergies areas were determined when high levels of PrES occur simultaneously with other services and/or biodiversity. For example, Thompson et al. (2009) reported that 76% of 21 different studies showed a direct relationship between the increase in forest biodiversity and an increase in carbon fixation. These interactions can be managed to reduce costs or to improve the multi-functionality of the managed landscape (Raudsepp-Hearne et al. 2010; Mori et al. 2017). For example, eco-friendly management practices such as silvopastoral systems at landscape level may improve with an integral management of the aesthetic values, protection against soil erosion, increase of long-term understory production and better biodiversity conservation (Peri et al. 2016b).

From results of the present work, we are able to identify those areas whit potential trade-offs and synergies between PrES and biodiversity at regional level and within the forest landscapes. In synthesis, we found the following main aspects:

(i) Humid steppes and shrublands were the most important areas to provided PrES (mainly sheep production), but also presented the major potential biodiversity values where plants, birds and darkling beetles presented medium to high PBM qualities. These areas occupied 19% of the total area of Santa Cruz Province, but less than 3% of these areas were under protection areas. Potential trade-off areas were identified (see Fig. 2.5, brown and orange hexagons), where traditional managements of sheep breeding in private lands (e.g. overgrazing, heterogeneous and large paddocks, and continuous grazing) can affect negatively the biodiversity of plants, birds (Peri et al. 2013, 2016c; Kusch et al. 2016) and darkling beetles (Carrara and Flores 2013). These negative impacts increase degradation processes (Del Valle et al. 1998; Gaitán et al. 2019) and eventually can decrease PrES supply (Peri et al. 2016c).

(ii) Dry steppes provided medium PrES (mainly from sheep breeding and oil production) with medium potential biodiversity values (lizards had the highest PBM quality). These areas occupied 66% of the total province, but scarcely represented the protected areas (less than 3% are protected). We located most of the potential different interaction between PrES and potential biodiversity (see Fig. 2.5), e.g. potential trade-offs (brown and orange hexagons) occurred in specific areas near the seacoast (e.g. livestock) and inside provincial reserves (e.g. due to oil extractive activities). Also, we identified areas that suggest new protected areas (dark green and green hexagons) in the central part of the province.

(iii) Sub-Andean grasslands, native forests and alpine vegetation occupied 17% of the total province and provided less PrES with low potential biodiversity. Forests and alpine vegetation occupied 7% of the Santa Cruz Province; however, these ecosystems are well represented as protected areas (75% protected). Because of this, we considered that these areas had low probability of potential trade-offs.

(iv) Forest landscape matrix presented important values of PrES and potential biodiversity, depending on forest types. Sheep production and silvopastoral systems had the major values when grasslands and NA forests were combined, or when NA forests prevailed. Timber production presented exceptional values when NP forest type was combined with NA or occurred close to grasslands. However, NA forests that represented 41% of the *Nothofagus* forest with exceptional potential biodiversity values were scarcely represented as protected areas (only 16% protected). In contrast, NP and mixed evergreen forests (represented 55% and 4% of the *Nothofagus* forest, respectively) are protected as national parks and provincial reserves (69% and 82%, respectively). We located the biggest area with potential trade-offs in the south, where NA forests prevail (see Fig. 2.5bV, brown and orange hexagons). In this context, areas that suggest new protected areas (dark green and green hexagons) are located in the north where NP is combined with NA forests (see Fig. 2.5bI, II, III). Several potential trade-offs between PrES and biodiversity occurred in NA forests in private lands without any protection and few regulations for conservation.

Landscape analyses allowed us to compare PrES and potential biodiversity for different ecological areas at regional level, as it was also reported in forests landscape at Tierra del Fuego Province, Argentina (Martínez Pastur et al. 2017). The intensification of livestock and forest harvesting without any consideration of other ES and biodiversity can affect the resilience of natural ecosystems (Cardinale 2012; Lindenmayer et al. 2012) as well as biodiversity values (MEA 2005; Mori et al. 2017). The importance of ES and biodiversity conservation incentivises public and private sectors to incorporate these concepts into decision-making (De Groot et al. 2010; Koschke et al. 2012). Recently, scientific and policy agendas on biodiversity have included evaluations of ES by incorporating a monitoring system to determine the effectiveness and progress of implemented public policy (Braat and De Groot 2012; Costanza et al. 2017). For this, it is necessary to consider multiple factors (De Groot et al. 2010), where the characterization and location (e.g. mapping) of ES and biodiversity are necessary to support decision-making at landscape scale (Raudsepp-Hearne et al. 2010; Cordingley et al. 2016; Turkelboom et al. 2018).

At global scale, different advances in economic valuation (De Groot et al. 2012), social perception (Reyers et al. 2013; Quintas-Soriano et al. 2016), conservation planning (Cordingley et al. 2016) and landscape planning (Koschke et al. 2012) for ES maintenance and biodiversity conservation have been developed. In this context, different studies tried to understand these interactions (trade-off and synergies) using different approaches in Argentina, particularly in Patagonia – for example, valuation of PrES from a socio-economic perspective (Laterra et al. 2011), ES provided by different managed ecosystems (Chillo et al. 2018; Rositano et al. 2018) or analyses of several trade-offs in productive ecosystems (e.g. silvopastoral) (Oñatibia et al. 2015; Martínez Pastur et al. 2017; Peri et al. 2016a). However, the challenge to solve trade-offs in the practice still remains. Understanding these relationships and creating maps that link PrES and biodiversity facilitate the connection of main society interests with natural ecosystems (Raudsepp-Hearne et al. 2010; Cordingley et al. 2015).

Our results allowed to (i) obtain empirical information about the provision of different PrES, (ii) define geographical distribution of PrES and potential biodiversity, (iii) identify hot and cold-spot areas, (iv) locate potential trade-off areas between different economic activities and biodiversity conservation values, (v) define areas to suggest new protected areas based on high values of PBM and low values of PrESM and (vi) define areas where the maximization of PrES can reduce ES losses. Moreover, with the identification of these areas of interest, it is possible to promote a balance between management and conservation strategies in private lands and develop new proposals for sustainable management at landscape level (e.g. variable retention harvesting) (Martínez Pastur et al. 2019) and also to contribute in public policies by improving the current management practices on private lands. For example, Law No. 26,331 promotes the use of native forests in a sustainable way to maintain their biodiversity and ecosystem services (Article 5). This law contemplates different uses in conservation categories to manage native forests looking for a balance of different ES provision (Peri and Ormaechea 2013; Peri et al. 2019a). In addition, the “National Plan for the Management of Forests with Integrated Livestock (MBGI)” has defined different guidelines for livestock and forestry activities under the maintenance of the structural and functional components of the native forest and therefore its ecosystem services (Peri et al. 2016b). In this context, our maps can be a powerful tool to develop land use, management and conservation proposals, based on multi-functionality of natural ecosystems (De Groot et al. 2010; Koschke et al. 2012; Maes et al. 2012).

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