

Linking potential biodiversity and three ecosystem services in silvopastoral managed forest landscapes of Tierra del Fuego, Argentina

Guillermo Martínez Pastur, Pablo Luis Peri, Alejandro Huertas Herrera, Stefan Schindler, Ricardo Díaz-Delgado, María Vanessa Lencinas & Rosina Soler

To cite this article: Guillermo Martínez Pastur, Pablo Luis Peri, Alejandro Huertas Herrera, Stefan Schindler, Ricardo Díaz-Delgado, María Vanessa Lencinas & Rosina Soler (2017) Linking potential biodiversity and three ecosystem services in silvopastoral managed forest landscapes of Tierra del Fuego, Argentina, International Journal of Biodiversity Science, Ecosystem Services & Management, 13:2, 1-11, DOI: [10.1080/21513732.2016.1260056](https://doi.org/10.1080/21513732.2016.1260056)

To link to this article: <https://doi.org/10.1080/21513732.2016.1260056>



© 2016 The Author(s). Published by Informa UK Limited, trading as Taylor & Francis Group



[View supplementary material](#)



Published online: 27 Dec 2016.



[Submit your article to this journal](#)



Article views: 1276



[View Crossmark data](#)



Citing articles: 5 [View citing articles](#)

Linking potential biodiversity and three ecosystem services in silvopastoral managed forest landscapes of Tierra del Fuego, Argentina

Guillermo Martínez Pastur ^{a,b}, Pablo Luis Peri ^{b,c}, Alejandro Huertas Herrera ^a, Stefan Schindler ^{d,e}, Ricardo Díaz-Delgado ^f, María Vanessa Lencinas ^{a,b} and Rosina Soler ^{a,b}

^aCentro Austral de Investigaciones Científicas (CADIC), Ushuaia, Argentina; ^bConsejo Nacional de Investigaciones Científicas y Técnicas (CONICET), Buenos Aires, Argentina; ^cInstituto Nacional de Tecnología Agropecuaria (INTA), Universidad Nacional de la Patagonia Austral (UNPA), Río Gallegos, Argentina; ^dDepartment of Conservation Biology, Vegetation & Landscape Ecology, University of Vienna, Vienna, Austria; ^eBiodiversity & Nature Conservation, Environment Agency Austria, Vienna, Austria; ^fLaboratorio de SIG y Teledetección, Estación Biológica de Doñana (CSIC), Sevilla, Spain

ABSTRACT

Several studies confirm that biodiversity loss endangers ecosystem services (ES) supply and human well-being. A better understanding of biodiversity–ES relationships and effects of biodiversity loss on ES supply is needed. The objective was to determine relationships between potential biodiversity and three ES in Patagonia where cattle ranching under silvopastoral use occurs. We used grids of potential biodiversity (plant species richness) and three ES, provisioning (cattle stocking rate), regulating (CO₂ sequestration) and cultural (geo-tagged digital-images). Potential biodiversity was negatively related to provisioning, but no significant relations were detected with regulating and cultural. These relations showed regional differences related to forest landscape distribution. High values of regulating were found in southern areas being coincident with high potential biodiversity. Opposite trends (negative relationship with biodiversity) was observed for provisioning in eastern and western regions where provisioning decrease from N-S. Results suggest that provisioning do not overlap spatially with the higher values of potential biodiversity maps, which is an advantage for land use planning when conservation and management requirements must be combined. Our results are the first contribution for Patagonia to underpin scientific and institutional efforts to connect biodiversity conservation with ES maintenance. However, further studies must be addressed including more ES and regions.

ARTICLE HISTORY

Received 30 May 2016
Accepted 6 November 2016

EDITED BY

Sandra Luque

KEYWORDS


Biodiversity conservation; ecosystem service supply; provisioning; regulating; cultural; trade-offs; silvopastoral management; sustainable practices


1. Introduction

The concept of ecosystem services (ES) has recently received increasing attention because its capacity to bridge connections between natural ecosystems and human society (Reyers et al. 2013), as well as to integrate ecological, socio-cultural and economic approaches with knowledge building capacity and policy development (de Groot et al. 2010; Martínez Pastur et al. 2016a). Understanding and quantifying constraints to multiple ES delivery and biodiversity is vital for developing management strategies for current and future human well-being (Maskell et al. 2013; Schindler et al. 2014, 2016). Several studies agree that the loss of biodiversity can endanger ES provision and human well-being (Díaz et al. 2006; Meli et al. 2014; Felipe-Lucia & Comín 2015; Soliveres et al. 2016) since species: (i) regulate many ecosystem processes and functions that underpin ES, (ii) are ES itself and (iii) operate as a good indicator subject to different kinds of valuation (Mace et al. 2012; Aslaksen et al. 2015). In this sense, biodiversity is assumed to be critical to ES supply (MEA 2005),

although an understanding of links between biodiversity and individual ES are weak (Kremen 2005; Isbell et al. 2011; Maskell et al. 2013; Jax & Heink 2015). Biodiversity and ES are both important for management and conservation targets. It is unclear to which extent there is an overlap between conservation objects when protecting ES and biodiversity, and to which extent ES replace or complement biodiversity conservation (Jax & Heink 2015). The understanding of the nature of biodiversity–ES relationship and the possible effects of biodiversity loss on the delivery of ES is therefore critical (Balvanera et al. 2006; Cardinale et al. 2006; Harrison et al. 2014).

The concept of ES consist of flows of materials, energy and information from natural capital stocks that combined with manufactured and human capital services produce human welfare sorted in four categories: provisioning, regulating, supporting and cultural services (Costanza et al. 1997; MEA 2005). In this context: (i) biodiversity supports critical provisioning ES for people, such as food and raw materials (MEA 2005; Loreau & de Mazancourt 2013); (ii)

CONTACT Guillermo Martínez Pastur  gpastur@conicet.gov.ar

 Supplemental data for this article can be accessed [here](#).

regulating ES have been associated to several biodiversity attributes (e.g. pest regulation) (Harrison et al. 2014) and play a role of ecosystem processes regulator (Mace et al. 2012); (iii) supporting ES have been directly connected to biodiversity, such as primary production or nutrient cycling of natural ecosystems (Zhao & Running 2010; Maskell et al. 2013; Harrison et al. 2014) and (iv) cultural ES are clearly linked with biodiversity though increasing well-being and satisfaction caused by aesthetic values based on emblematic species and habitats (Díaz et al. 2006; Sandifer et al. 2015; Martínez Pastur et al. 2016a).

Policies to promote ES supply while safe-guarding biodiversity conservation are increasingly demanded by researchers, land managers and society (Haines-Young & Potschin 2010; Felipe-Lucia & Comín 2015). Managing natural ecosystem requires the analysis of trade-offs and synergies between conservation and economic efforts (Tallis et al. 2008). Several recent studies revealed cost-effective management schemes to enhance ES provision while protecting biodiversity (Felipe-Lucia & Comín 2015). Lands aimed at agroforestry, e.g. areas under permanent pastures combining forestry and grazing, offer provisioning ES vital for human well-being and constitute a key component of the global economy (Millennium Ecosystem Assessment (MEA) 2005; McCouch et al. 2013; Duru et al. 2015). These agroforestry land-use systems greatly increased during the last years, e.g. global food production increased 2.5 times from 1960 to 2000 (Millennium Ecosystem Assessment (MEA) 2005), leading to a land use intensification with loss of crucial natural and semi-natural habitats (Foley et al. 2005) on which biodiversity is depending (Tilman et al. 2001; Duru et al. 2015). However, agroforestry systems provide advantages compared with conventional agricultural and forest production methods due to better biodiversity conservation and multiple ES supplies (Fischer et al. 2011; Durán et al. 2014). Agroforestry provides multiple ES and combines the provision of animal and forestry products with non-commodity outputs, such as climate, water and soil regulation, and recreational, aesthetic and cultural heritage values (McAdam et al. 2009; Smith et al. 2013; Fagerholm et al. 2016).

Usually, human welfare is measured in Tierra del Fuego (Argentina) by traditional provisioning ES: provide forage and habitat for cattle, or wood from harvesting (Peri et al. 2016; Martínez Pastur et al. 2016b). However, Fuegian landscapes also host grasses, forbs, shrubs and trees in various communities that provide habitat to many native and several exotic species. Here, the relationship between biodiversity and ES is still poorly understood (Martínez Pastur et al. 2016a), unlike in other parts of the world some of these relationships were identified (e.g. Harrison et al. 2014; Bennett et al. 2015; Cordingley

et al. 2016). Thus, the aim of this research was to identify relationships between biodiversity and ES in the central part of Tierra del Fuego (Argentina) dominated by agroforestry systems. For this reason, we analyse the relationships between plant species richness as the most often used surrogate for biodiversity and three representative ES of provisioning, regulating and cultural.

2. Methods

The study was carried out in the central area of Tierra del Fuego province located at Southern Patagonia (Argentina) (53°30' to 54°30' S, 66°30' to 68°36' W) covering 5,053 km². A rainfall gradient from north (dry) to south (wet) defines the main vegetation units: the landscapes dominated by steppe of native grasses species in the north and the forest dominated landscapes in the south. For this study, the forest landscapes dominated by *Nothofagus antarctica* (G. Forst.) Oerst was selected (Figure 1(a)). This tree species can growth up to 14 m height in high-quality sites, or as shrubs of 1.3 m height in xeric sites (Ivancich et al. 2014). The *N. antarctica* forests cover 1.820 km² in Tierra del Fuego (Collado 2001) and are mainly used for agroforestry purposes (silvo-pastoral systems) where private ranches have been established in a transitional area with the Fuegian steppe (Peri et al. 2016).

The relation between potential biodiversity and the three selected ES was analysed by a comparison of grid maps with estimated values for each studied variables (Schneiders et al. 2012). We divided the study area in squares of 3.1 km², resulting in 163 sampling units or windows. These units were defined using ArcGIS 9.3 software (ESRI 2008) and the Euclidean allocation of spatial analyst tool. Beside this, we assigned each sampling unit into a regional reference framework along N-S and W-central-E gradients, thus six combinations resulted (Figure 1(c)). The studied gradients were determined by different drivers: (i) the northern area is influenced by a high proportion of steppe habitats while the southern area by a high proportion of forest habitats (Figure 1(a)); (ii) the W-central-E gradient includes a climate gradient, with temperatures increasing and rainfall decreasing from west to east due to the influence of the Atlantic Ocean in the east (Kreps et al. 2012). The eastern region has also higher human accessibility since the only national route and most of secondary roads go through it (Figure 1(b)). In each sampling window we analysed the supply of provisioning, regulating, and cultural services using one service per group; and we characterized the potential biodiversity through specific developed maps (Martínez Pastur et al. 2016c, see Supplementary material, Annex 1 and 2): (i) For the provisioning ES we used a map of above-ground biomass of the understory species growing in N.

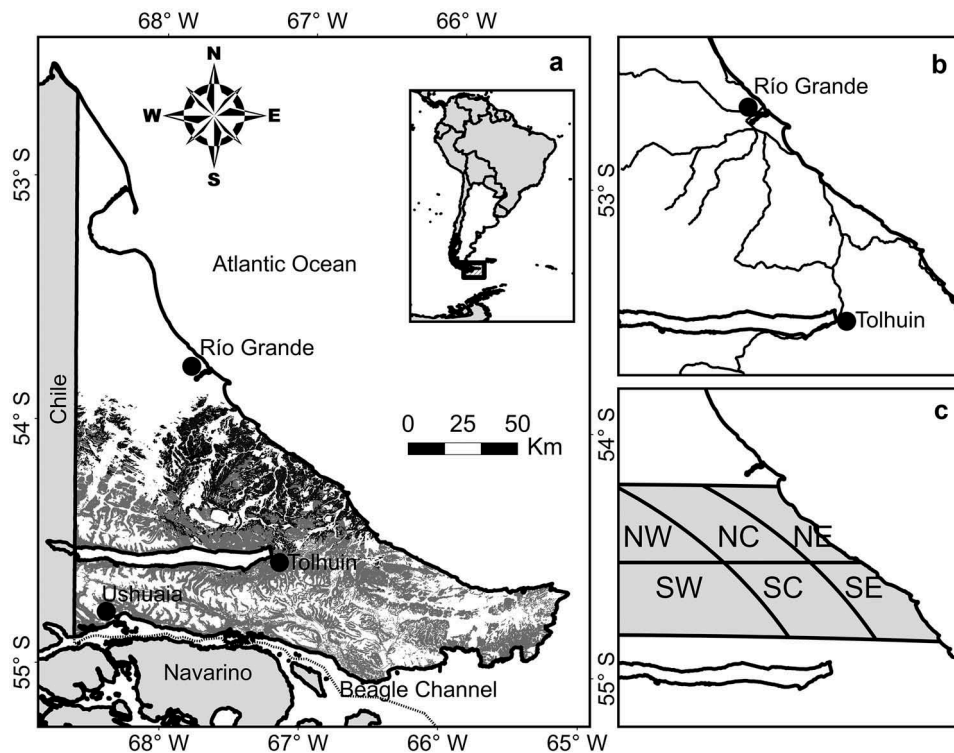


Figure 1. Study area in Tierra del Fuego (Argentina): (a) location, cities and forest types (dark grey = *Nothofagus antarctica*, pale grey = *N. pumilio* and *N. betuloides*); (b) national and provincial routes and (c) study area and defined regions (N = north, S = south, W = west, C = central, E = east).

antarctica forest landscapes ($\text{kg dry matter}\cdot\text{ha}^{-1}\text{ yr}^{-1}$) defined for each forest unit (Peri 2009). Based on this, we calculated the average cattle stocking rate (Holechek 1988) referred to the above-ground biomass allowance of the understory (bovine equivalent, $\text{ind}\cdot\text{ha}^{-1}\text{ yr}^{-1}$) according to Borrelli (2001). (ii) For regulating ES we used a map of average carbon fixation of *N. antarctica* forests. We calculated the average forest carbon fixation of each sampling window ($\text{ton}\cdot\text{ha}^{-1}\text{ yr}^{-1}$) using CO_2 sequestration as a surrogate of gas regulation (Trabucchi et al. 2014; Felipe-Lucia & Comín 2015). For this, we estimated the carbon fixation using the global MODIS net primary productivity (NPP) algorithm (Running et al. 2004) using spatially explicit NPP of the year 2014 based on Zhao & Running (2009) algorithm. Briefly, data were obtained from MODIS sensor collection 5 (C5) 8-day composite 1-km fraction of photosynthetically active radiation (FPAR) and leaf area index (LAI). Then, pixels were classified according to the *N. antarctica* forest cover based on Collado (2001) and Allué et al. (2010) maps. (iii) For cultural ES we quantified geo-tagged digital images of the study area that local people and visitors posted on the Panoramio web platform (Martínez Pastur et al. 2016a). We quantified and mapped the photos considering the social and biophysical importance of four different cultural services (aesthetic value, existence value, recreation and local identity). We merged all the cultural services types in only one category for the further analyses and calculated the density

of photos for each sampling window (photos per 10 thousand hectares). (iv) A map of plants species richness as surrogate of potential biodiversity (MPB) was recently developed for *N. antarctica* forests (see methods and main outputs in Martínez Pastur et al. 2016c). In brief, using Ecological Niche Factor Analysis (Hirzel et al. 2002) a series of spatially explicit habitat suitability models was produced for 20 vascular understory plants in Biomapper 4.0 software (Hirzel et al. 2004). These models were generated from species-presence data collected in 535 surveys and 15 selected environmental explanatory variables. The MPB resulted from the combination of the 20 species specific habitat suitability maps (HSM) (Hirzel & Arlettaz 2003), where species had been selected based on their importance (occurrence + cover) in *N. antarctica* forests (Martínez Pastur et al. 2016c). The MPB showed in each pixel the average HSM values of the 20 understory plant species as an indicator of potential biodiversity of the site (0 minimum and 100 maximum potential biodiversity).

MPB values and the studied ES data were integrated into a Geographic Information System (GIS) based on ArcGIS 9.3 software (ESRI 2008), contained unsigned 8-bit values and a resolution of 90 m. All files were projected in the World Geodetic System 1984 coordinates (WGS 84). The average value of each ES and the MPB scores per grid cell were calculated with the zonal statistics tool of the spatial analyst extension in ArcGIS 9.3 software (ESRI 2008). This tool calculates the average of all cells in the grid

that belong to the same sampling window (Schneiders et al. 2012), and in order to deal with the proposed objectives we conducted the following analyses: (i) a two-way ANOVAs comparing two levels of potential biodiversity (low and high) and the six regions (NE, SE, NC, SC, NW, SW) as predictors for each ES type (provisioning, regulating, cultural) as response variables, where mean comparisons were conducted by the Tukey test ($p < 0.05$); (ii) a one-way ANOVAs comparing potential biodiversity and ES types (provisioning, regulating, cultural) along the studied gradients (N-S and W-central-E) and Tukey tests to compare the mean values ($p < 0.05$); (iii) a principal components analysis (PCA) to determine the association between the two levels of potential biodiversity (low and high) with each ES type (provisioning, regulating, cultural) and (iv) to test the influence of the landscape on the relationship between each ES provision and potential biodiversity, a tendency graph was constructed comparing potential biodiversity and the different ES types (provisioning, regulating, cultural) along the studied gradients in the six geographically defined regions (cf. Figure 1(c)). We used PC-Ord software for the PCA (McCune & Mefford 1999). Only three components were generated in the PCA because three variables were utilized (the three studied ES types). The two first components were retained because explain 74% cumulative variance. The significance of the eigenvalues in the PCA was tested with Monte Carlo permutation test with 999 randomizations. Rotations were not applied.

3. Results

Plant species richness as surrogate of mean potential biodiversity values showed significant inverse relations with the provisioning service of above-ground understory species biomass ($p = 0.001$). Potential biodiversity did not relate with the supply of regulating and cultural ES throughout the regions (Table 1). The supply of regulating ES across the defined regions was significantly greater ($p < 0.001$) in southern than northern forests, while provisioning ES marginally changed among regions ($p = 0.056$) being greater in western than eastern forests. The supply of cultural ES did not change across the defined regions (Table 1 and Figure 2).

These trends were also detected in the multivariate analysis (Figure 3). We observed an overlap between sampling windows with lower and higher values of potential biodiversity, although the low values of potential biodiversity group showed greater dispersion than the high-values group (Figure 3). Axis 1 (Eigenvalue = 1.225, explained variance = 40.8%) separated the sampling windows according to the supply of regulating and cultural ES (Eigenvector of 0.69 and 0.71, respectively).

Table 1. Effect of plant species richness as surrogate of potential biodiversity level (high-low) and the geographical region of the study area within the distribution of *N. antarctica* forest landscapes on the provision of ecosystem services (CES = cultural, RES = regulating, PES = provisioning, N = north, S = south, E = east, C = central, W = west) in Tierra del Fuego, Argentina.

Predictor level		CES	RES	PES
A: Potential biodiversity	High	2.12	4.04	0.31a
	Low	5.84	3.98	0.42b
	F(p)	0.74 (0.397)	0.19 (0.660)	11.62 (0.001)
B: Region	NE	4.67	4.10b	0.34
	SE	11.78	4.62c	0.33
	NC	0.37	3.64ab	0.31
	SC	3.80	4.13b	0.39
	NW	2.47	3.58a	0.42
	SW	0.86	4.00ab	0.41
	F(p)	1.40 (0.228)	14.61 (<0.001)	2.22 (0.056)
Interaction: A x B	F(p)	1.37 (0.242)	1.02 (0.409)	1.11 (0.359)

F = F values (Fisher test); p = probability. Different letters showed statistically significant differences based on Tukey tests ($p < 0.05$).

Especially the last one was highly influenced by one hot spot occurring in the study area, the well-known touristic and recreation site San Pablo cape in Tierra del Fuego (Martínez Pastur et al. 2016a). Axis 2 (Eigenvalue = 1.002, explained variance = 33.4%) separated the sampling windows according to the supply of provisioning ES values (Eigenvector = 0.97). However, the Eigenvalues in the PCA for axis 1 and 2 was not significant ($p = 0.081$ and 0.425, respectively).

We detected some influence of the landscape over the provision of the selected ES and plant species richness as surrogate of potential biodiversity by influencing the relationship among them. Thus, potential biodiversity significantly changed in most of the studied gradients (Figure 4). N-S significantly differed in central ($p < 0.001$) and E ($p < 0.001$) areas, while no difference was found in W area ($p = 0.366$). Higher values of potential biodiversity were found in S compared with N areas. Furthermore, W-central-E significantly differed in N ($p < 0.001$) and S ($p = 0.012$) areas. In general, higher values were found in W than E areas, where central area presenting intermediate values.

Cultural ES did not significantly changed across the studied gradients ($p > 0.129$), except for a single case (Figure 4) where W-central-E showed significant differences in the northern area ($p = 0.002$), because greater values of cultural ES were found in E than in central-W areas. Regulating ES significantly changed across the studied gradients (Figure 4). If we consider the W-central-E gradient, N-S significantly differed in W ($p = 0.004$), central ($p < 0.001$) and E ($p = 0.007$) areas. Higher values of regulating ES were found in S than in N areas. If we consider the N-S gradient, W-central-E significantly differed in N ($p < 0.001$) and S ($p < 0.001$) areas. Higher values of regulating ES were found in W-central than in E areas. Finally, provisioning ES did not significantly changed across the studied gradients

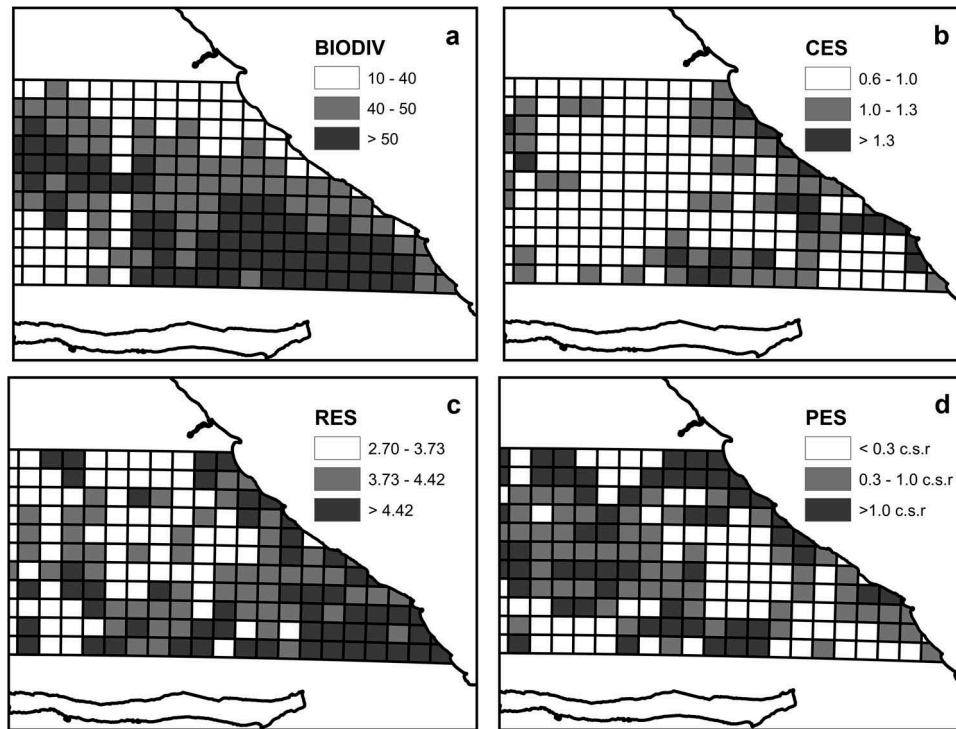


Figure 2. Average values of the sampling windows in the studied area: (a) plant species richness as surrogate for potential biodiversity (BIODIV) (based on Martínez Pastur et al. 2016c); (b) cultural ecosystem services (CES) estimated through the number of geo-tagged digital images (photos per 10,000 ha) (based on Martínez Pastur et al. 2016a); (c) regulating ecosystem services (RES) estimated through the carbon fixation rate of *N. antarctica* forests ($\text{ton}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) (based on Zhao and Running 2009) and (d) provisioning ecosystem services (PES) estimated through the cattle stocking rate (bovine equivalent, $\text{ind}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) (based on Peri 2009).

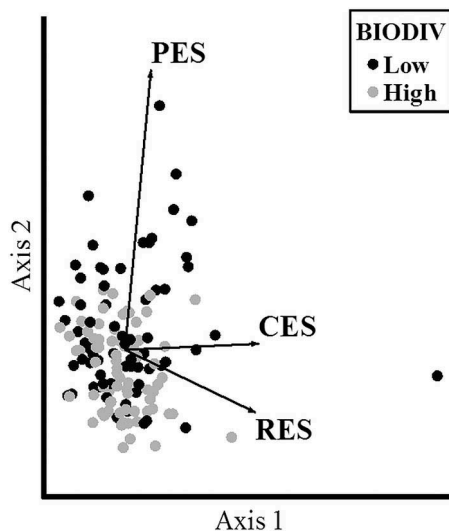


Figure 3. Multivariate (PCA) analysis of ecosystem services (CES = cultural, RES = regulating, PES = provisioning) for two categories of plant species richness as surrogate for potential biodiversity (BIODIV).

(all p -values > 0.075), except for a single case (Figure 4) where N-S significantly differed in the eastern part of the study area ($p = 0.005$), because greater values of provisioning ES were found in N than in S areas.

4. Discussion

4.1. Relation of potential biodiversity and the supply of the three selected ES

Potential biodiversity and ES supply of the selected variables significantly changed at landscape level, being associated with different climatic, biophysical and social factors. The potential biodiversity in *N. antarctica* forests was greater than in other *Nothofagus* forests. For instance, the species occurrence in these forests varies along their latitudinal distribution range and habitat conditions, with different characteristic species from N to S, and from xeric to humid environments (Lencinas et al. 2008; Peri et al. 2016). Also, economic human-related activities induced changes in biodiversity through harvesting and livestock grazing in silvopastoral systems (Peri et al. 2016). The map of potential biodiversity for the study area was built using habitat suitability models of understory plants, based on climate, topographic and forest variables (Martínez Pastur et al. 2016c). Plant species that occurred in northern areas (areas with dry conditions and high mean temperatures) presented lower marginality values, while plant species in southern areas (high rainfall and lower mean temperatures) had higher marginality. In the studied forests, plant

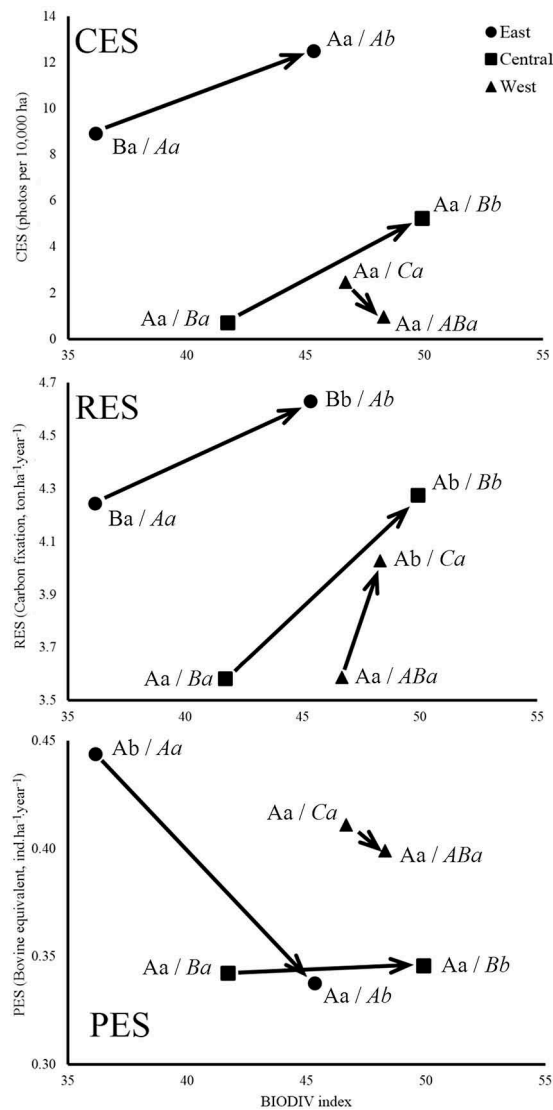


Figure 4. Provision of ecosystem services (CES = cultural, RES = regulating, PES = provisioning) related to potential biodiversity (BIODIV) along the regions (W = west, central, E = east) where arrows showed the N-S (N = north, S = south) pattern. Differences of Tukey test ($p < 0.05$) were presented in upper cases for W-central-E comparisons for each N-S regions, and lower cases for N-S comparisons for each W-central-E regions. In the left side results of Y axes were presented (CES, RES, PES) and in the right side (italic cases) results of X axes were presented (BIODIV).

Significance (F , p) of ANOVAs in north-south comparisons were: East = BIODIV (24.71, <0.001), CES (0.23, 0.634), RES (7.99, 0.007), PES (8.66, 0.005); Central = BIODIV (45.64, <0.001), CES (2.37, 0.130), RES (25.99, <0.001), PES (0.01, 0.923); and West = BIODIV (0.83, 0.366), CES (1.42, 0.240), RES (9.17, 0.004), PES (0.14, 0.711). Significance (F , p) of ANOVAs in west-central-east comparisons were: North = BIODIV (27.13, <0.001), CES (6.96, 0.002), RES (14.99, <0.001), PES (2.55, 0.086); and South = BIODIV (4.62, 0.013), CES (1.82, 0.169), RES (9.24, <0.001), PES (2.62, 0.079).

species are more specialized than in other forest types of Tierra del Fuego Island by presenting narrower environmental niches closely related to climate (e.g. rainfall gradient and soil moisture availability), topographic (e.g. altitude and exposition) and forest structural variables (e.g. crown cover that was closely

related to light availability inside the forest) (Lencinas et al. 2008; Peri et al. 2016).

In the study area, the supply of cultural ES is influenced by the closeness to water bodies (Martínez Pastur et al. 2016a) due to aesthetic reasons (García-Llorente et al. 2012) and recreation activities (Abildtrup et al. 2013) mainly in the marine coastline. Also particular vegetation types supplied a higher level of cultural ES (Martínez Pastur et al. 2016a) and it can be interpreted as an expression of phytophilia (García-Llorente et al. 2012; López-Santiago et al. 2014) determining more preferences for forests than grasslands ecosystem. However, local identity in Tierra del Fuego increased in grasslands due to human activities related to ranching activities. In the study area, human population is scarce, and visitors appreciate signs of civilization (e.g. ranch constructions or shipwrecks) which represents hot spots of cultural ecosystem services (CESs) (e.g. San Pablo cape). In addition, the accessibility is crucial for the supply of cultural ES. Areas with high levels of cultural ES are coincident with good accessibility (paved roads), as it was described in previous studies (e.g. Abildtrup et al. 2013; Richards & Friess 2015).

Usually, the supply of regulating ES is related to intermixed relationships among biodiversity, ecosystem types, climate and topography (Harrison et al. 2014; Trabucchi et al. 2014; Felipe-Lucia & Comín 2015). We choose as indicator for this ES group the CO₂ sequestration (Running et al. 2004) that has been previously proposed by several authors (e.g. Trabucchi et al. 2014; Felipe-Lucia & Comín 2015). In this context, forest ecosystem pools and fluxes of C were strongly affected by historical land use (e.g. wood extraction and livestock pressure). This resulted in forests with different structures determined by changes in the proportion of crown classes, stand development stages (age) and the site quality where trees grown (Peri et al. 2010; Peri 2011; Kreps et al. 2012). In our study case, forests growing in higher site quality and greater C pools are located in the ecotone with *N. pumilio* forests at the southern study area.

Finally, provisioning ES in the study area are mainly related with the supply of plant forage for cattle grazing and supply of biotic materials in form of wood harvesting (Peri et al. 2016). Cattle ranching is based on natural forage species growing under the forest canopy. The productivity and nutritive value of these forage (natural grasses) are dependent on the interaction of environmental (mainly soil water availability and light intensity) and management factors under the trees which determine the cattle performance (Peri et al. 2016). Cattle stocking rate changed across the landscape based on the rainfall gradient decreasing from N-E to S-W (Kreps et al. 2012) and light availability under the tree canopy, where high-quality site stands (greater height trees) or unmanaged stands provide lower light quantities inside

the forests (Peri 2009; Peri et al. 2016), and consequently less understory grasses. Timber is used by rural carpenters and fire wood, and it is still important in the area, although fossil gas is the main heating resource for local inhabitants (Gea et al. 2004).

Several studies showed positive effect of biodiversity on ES supply (Quijas et al. 2010; Duru et al. 2015; Soliveres et al. 2016). Biodiversity regulates ecosystem processes and determines delivery of several ES, where the adequate combination of biotic and abiotic components must occur at any particular place and time (de Groot et al. 2010; Mace et al. 2012). In the studied area, plant species richness as surrogate for potential biodiversity was poorly linked with the studied cultural ES values. This is mainly because the study area is a non-touristic place with low accessibility, and landscape perceptions are more important than existence values (Martínez Pastur et al. 2016a). Contrary to our findings for Tierra del Fuego, many studies about biodiversity linkages with cultural ES described positive relations (Hough 2014; Sandifer et al. 2015).

In the studied area, potential biodiversity was also poorly related to the studied regulating ES. CO₂ sequestration showed a strong relation with climate gradients, which in Tierra del Fuego are defined by annual rainfall and mean temperature (Peri et al. 2010; Kreps et al. 2012), however many other climatic factors also have great importance over ecological processes that influence over tree growth and ecosystem dynamics (see Martínez Pastur et al. 2011; Peri 2011; Torres et al. 2015). The proximity of other habitats associated with *N. antarctica*, either open or forested lands, affects the potential biodiversity of *N. antarctica* by supporting populations of ecotone species in both forest-steppe or forest-forest ecotone areas (Lencinas et al. 2008; Martínez Pastur et al. 2016c). Contrary to several other studies across the globe, reviewed by Harrison et al. (2014), we did not find evidence of associations between the studied regulating ES and biodiversity attributes for our Fuegian study area. Lastly, potential biodiversity was inversely linked with provisioning ES values, because higher cattle stocking rate occurred in the less valuable *N. antarctica* forest landscapes in terms of potential biodiversity. While forests with complete crown cover intermixed with other forested lands (close forest-forest ecotone) are the better areas for biodiversity, the open forests near sea shores (close forest-steppe ecotone) are the best zones for livestock production (Lencinas et al. 2008; Peri 2009; Peri et al. 2016; Martínez Pastur et al. in press). Cordingley et al. (2016) also found that biodiversity value and provisioning ES were related to ecosystem characteristics in England (e.g. size of heathland patches). This indicates that trade-offs can occur between different ES, and between ES and biodiversity indicators (e.g.

carbon storage, timber, aesthetic value, recreation) (Schindler et al. 2014).

4.2. Potential biodiversity and the supply of ES in agroforestry landscapes

Several studies showed how biodiversity attributes affecting provision of ES by impacting the underlying ecosystem processes (Díaz et al. 2006; Harrison et al. 2014). These studies were conducted at species level, group of species or at broader scale, analysing biodiversity impact over a single or multiples ES (Kremen 2005; Díaz et al. 2007; Luck et al. 2009; Poirazidis et al. 2011; Harrison et al. 2014). In Tierra del Fuego, studies that relate biodiversity values and ES supplies have only been performed at stand level (e.g. Lencinas et al. 2008), whereas our study is the first work that approach this type of analyses at landscape level, considering the study of some ES supply. The relationship between biodiversity and ES is often complex. The relations between biodiversity and ES are direct or inverse largely depending on the type of ES (Mace et al. 2012), where direct relations are more often detected with pollination services or cultural services (Lucas et al. 2014; Martínez Pastur et al. 2016a). Beside this, species populations are the key unit in the supply of ES (Luck et al. 2003) because biodiversity is the factor on which ecosystems run and sustain their functions with a consequent impact on human economy (Deliège & Neuteleers 2015). In our study area, livestock production (80% cattle, 20% sheep) is the main annual income from silvopastoral systems. Animal performance at the whole farm scale is presented by comparing traditional extensive grazing management with an adaptive silvopastoral management that includes strategic separation in homogeneous areas, stocking rate adjustment to forage availability (net primary production) and the protection of tree regeneration against animal (domestic or native) over-browsing (Peri 2009; Peri et al. 2016). Silvopastoral management in *N. antarctica* forests maintained most of the original pastures characteristics. Also, the native herbivore populations (e.g. *Lama guanicoe*) use the same niches than cattle and sheep (Martínez Pastur et al. 2016b). The challenge for sustainable management production is to apply technical solutions while avoiding the impairment of other ES, especially regulating ES. In fact, the capacity to maintain these provisioning ES are often maintained and enhanced by technical means (Lucas et al. 2014), e.g. irrigation, livestock management or seeding old pastures with more palatable forage species. Appropriate technical solutions remain difficult, especially for owners with few economic resources (Foley et al. 2011; Lucas et al. 2014). Furthermore, technical solutions can have negative impact on the supply of regulating and cultural ESs as shown by

Schindler et al. (2014) for European floodplains under multiple uses, including agriculture. Several authors suggested that maintaining high levels of plant diversity in agro-forest ecosystems shows a clear positive effect over provisioning ES (Quijas et al. 2010; Duru et al. 2015). Also, maintaining heterogeneous landscapes, such as occurring in Tierra del Fuego, allows having high spatiotemporal biodiversity from small patches to the entire landscape level, increasing the chance of recovery of the associated biodiversity after any kind of impact (Duru et al. 2015). This provides resilience and stability of ecological processes in changing environments (Tschardt et al. 2012; Geijzendorffer & Roche 2013), where biodiversity appears to regulate ecosystem processes and determine ES supply in agro-forest landscapes.

5. Conclusion

The intensification of agriculture and agroforestry has already shown that optimization of one or some ES (e.g. provisioning) are likely to reduce diversity and system stability (Cardinale et al. 2012; Lindenmayer et al. 2012), as well as biodiversity (MEA 2005). Recently, science and policy agendas on biodiversity moved to include ES assessments and recognized the crucial task of monitoring ES for determining the effectiveness and progress of policy frameworks (Geijzendorffer & Roche 2013; Liqueste et al. 2016). Understanding the processes behind forest ES provision, as well as their trade-offs with biodiversity conservation, is a useful tool to support spatial planning and land management (Poirazidis et al. 2011; Carvalho-Santos et al. 2015). Trade-off between agroforestry practices and biodiversity has been widely reported, with major implications for environmental management (Cordingley et al. 2016). Improved decision-making in land management relating to such trade-offs requires empirical information on the relationships between ecosystem management and provision of ES at the landscape scale (de Groot et al. 2010). Some previous studies have examined the impact of landscape-scale conservation management approaches on trade-offs between biodiversity and ES (Birch et al. 2010; Newton et al. 2012; Hodder et al. 2014; Cordingley et al. 2016), and the challenge still is how to solve such trade-offs in the practice. In this paper, we analyse these relationships at landscape level for silvopastoral management and potential biodiversity considering three monetary and non-monetary ES. These analyses assist to identify the optimal allocation for different management options along the *N. antarctica* distribution (De Groot et al. 2010), where the explicit consideration of trade-off choices should

itself lead to improve the conservation proposals (McShane et al. 2011; Cordingley et al. 2016). Productive system proposals in Tierra del Fuego generated few changes in the biodiversity, maintaining the original forest types (e.g. *N. antarctica*), the native fauna (e.g. large ungulates as *Lama guanicoe*) and most of the original understorey plants under the tree canopy (Peri 2009; Peri et al. 2016). The current proposal for sustainable silvopastoral systems (Peri et al. 2016) aims to manage these forests to enhance multiple ES supply while protecting biodiversity as also was proposed for other parts of the world by policies encouraging multifunctional landscapes (e.g. Felipe-Lucia & Comín 2015; Schindler et al. 2016).

Provisioning ecosystem services (PESs) did not overlap spatially with the higher values of potential biodiversity in the studied area. This provides an advantage for land use planning when conservation and management requirements must be combined. These areas with higher potential biodiversity are neither linked with the areas with highest supply of cultural ES in native *N. antarctica* forests, because visitors appreciate signs of civilization and accessibility far from the most diverse regions. Unfortunately, such unconnected forest–society relationship underestimate the biological legacy of these native forests. Our results are the first contribution for southern Patagonian forests that underpins scientific and institutional efforts in policies to connect biodiversity conservation with ES maintenance and enhancement its supply. More studies must be addressed for Tierra del Fuego including more ES to reach to definitive conclusions and develop new management alternatives. Incorporating traditional conservation strategies for species and habitat protection within the broader context of social-ecological systems and ES delivery can lead to added benefits for biodiversity through closer integration of conservation policy with policies in other sectors. To date, the strategy was based on maintaining and protecting areas of special interest (land sparing), however it is essential to improve management practices in productive areas to assure the delivery of multiple ES (land sharing). For developing these new management practices it is essential to quantify the synergies and trade-offs between biodiversity and ES provision that should be readily incorporated into decision support tools to foster better spatial planning of ES supply. Finally, it is necessary to consider that our study was conducted in a region with recent European human impact (c.a. 100 years), and further similar studies must be designed in other regions with greater environmental changes.

Disclosure statement

No potential conflict of interest was reported by the authors.

Funding

This research was supported by 'Operationalisation of ecosystem services and natural capital: From concepts to real-world applications' (OpenNESS) project financed under the European Commission's Seventh Framework Programme (Project number 308428). This paper was also written with the financial support of MINCYT-BMWF Cooperation Programme (2011-2013) through the project: 'Impact of variable retention management in *Nothofagus pumilio* forests of Southern Patagonia: A meta-analysis across 15 years of results for multiple taxa and scales'.

ORCID

Guillermo Martínez Pastur  <http://orcid.org/0000-0003-2614-5403>

Pablo Luis Peri  <http://orcid.org/0000-0002-5398-4408>

Alejandro Huertas Herrera  <http://orcid.org/0000-0003-2229-6714>

Stefan Schindler  <http://orcid.org/0000-0002-1755-4304>

Ricardo Díaz-Delgado  <http://orcid.org/0000-0002-0460-4616>

María Vanessa Lencinas  <http://orcid.org/0000-0002-2123-3976>

Rosina Soler  <http://orcid.org/0000-0002-5799-1672>

References

- Abildtrup J, Garcia S, Olsen SB, Stenger A. 2013. Spatial preference heterogeneity in forest recreation. *Ecol Econ*. 92:67–77.
- Allué C, Arranz JA, Bava J, Beneitez JM, Collado L, García-López JM. 2010. Caracterización y cartografía fitoclimáticas del bosque nativo subantártico en la Isla Grande de Tierra del Fuego (Patagonia, Argentina). *For Syst*. 19:189–207.
- Aslaksen I, Nybø S, Framstad E, Garnåsjordet PA, Skarpaas O. 2015. Biodiversity and ecosystem services: the nature index for Norway. *Ecosyst Ser*. 12:108–116.
- Balvanera P, Pfisterer AB, Buchmann N, He J, Nakashizuka T, Raffaelli D, Schmid B. 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecol Lett*. 9:1146–1156.
- Bennett E, Cramer W, Begossi A, Cundill G, Díaz S, Ego B, Geijzendorffer I, Krug C, Lavorel S, Lazos E, et al. 2015. Linking biodiversity, ecosystem services, and human well-being: three challenges for designing research for sustainability. *Curr Opin Environ Sustainability*. 14:76–85.
- Birch J, Newton AC, Alvarez Aquino C, Cantarello E, Echeverria C, Kitzberger T, Schiappacasse I, Tejedor Garavito N. 2010. Cost-effectiveness of dryland forest restoration evaluated by spatial analysis of ecosystem services. *Pnas*. 107:21925–21930.
- Borrelli P. 2001. Planificación del pastoreo. In: Oliva G, editors. *Ganadería sustentable en la Patagonia Austral*. Río Gallegos: Ed INTA-Centro Regional Patagonia Sur; p.183–196.
- Cardinale B, Srivastava D, Duffy J, Wright J, Downing A, Sankaran M, Jouseau C. 2006. Effects of biodiversity on the functioning of trophic groups and ecosystems. *Nature*. 443:989–992.
- Cardinale BJ, Duffy JE, Gonzalez A, Hooper DU, Perrings C, Venail P, Narwani A, Mace GM, Tilman D, Wardle DA, et al. 2012. Biodiversity loss and its impact on humanity. *Nature*. 486:59–67.
- Carvalho-Santos C, Sousa-Silva R, Goncalves J, Honrado JP. 2015. Ecosystem services and biodiversity conservation under forestation scenarios: options to improve management in the Vez watershed, NW Portugal. *Reg Environ Change*. doi:10.1007/s10113-015-0892-0
- Collado L. 2001. Tierra del Fuego forest: analysis of their stratification through satellite images for the province forest inventory. *Multequina*. 10:01–16.
- Cordingley JE, Newton AC, Rose R, Clarke R, Bullock J. 2016. Can landscape-scale approaches to conservation management resolve biodiversity-ecosystem service trade-offs? *J Appl Ecol*. 53:96–105.
- Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill R, Paruelo J, et al. 1997. The value of the world's ecosystem services and natural capital. *Nature*. 387:253–260.
- de Groot RS, Alkemade R, Braat L, Hein L, Willemen L. 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecol Complex*. 7:260–272.
- Deliège G, Neuteleers S. 2015. Should biodiversity be useful? scope and limits of ecosystem services as an argument for biodiversity conservation. *Environ Val*. 24:165–182.
- Díaz S, Fargione J, Chapin FS, Tilman D. 2006. Biodiversity loss threatens human well-being. *PLoS Biol*. 4:e277.
- Díaz S, Lavorel S, de Bello F, Quetier F, Grigulis K, Robson TM. 2007. Incorporating plant functional diversity effects in ecosystem service assessments. *Pnas*. 104:20684–20689.
- Duran AP, Duffy JP, Gaston KJ. 2014. Exclusion of agricultural lands in spatial conservation prioritization strategies: consequences for biodiversity and ecosystem service representation. *Proc R Soc B*. 281:1529.
- Duru M, Therond O, Martin G, Martin-Clouaire R, Magne MA, Justes E, Journet EP, Aubertot JN, Savary S, Bergez J, Sarthou JP. 2015. How to implement biodiversity-based agriculture to enhance ecosystem services: A review. *Agron Sustain Dev*. 35:1259–1281.
- [ESRI] Environmental Systems Research Institute. 2008. ArcGIS Desktop: Release 9.3. Redlands (CA): Environmental Systems Research Institute.
- Fagerholm N, Torralba M, Burgess PJ, Plieninger T. 2016. A systematic map of ecosystem services assessments around European agroforestry. *Ecol Ind*. 62:47–65.
- Felipe-Lucia M, Comín F. 2015. Ecosystem services-biodiversity relationships depend on land use type in floodplain agroecosystems. *Land Use Policy*. 46:201–210.
- Fischer J, Batary P, Bawa KS, Brussaard L, Chappell MJ, Clough Y, Daily GC, Dorrough J, Hartel T, Jackson LE, et al. 2011. Conservation: limits of land sparing. *Science*. 334:593.
- Foley JA, Defries R, Asner GP, Barford C, Bonan G, Carpenter SR, Chapin FS, Coe MT, Daily GC, Gibbs HK, et al. 2005. Global consequences of land use. *Science*. 309:570–574.
- Foley JA, Ramankutty N, Brauman KA, Cassidy ES, Gerber JS, Johnston M, Mueller ND, O'Connell C, Ray DK, West PC, et al. 2011. Solutions for a cultivated planet. *Nature*. 478:337–342.
- García-Llorente M, Martín-López B, Iniesta-Arandia I, López-Santiago CA, Aguilera PA, Montes C. 2012. The role of multi-functionality in social preferences toward semi-arid rural landscapes: an ecosystem service approach. *Environ Sci Pol*. 19–20:136–146.

- Gea G, Martínez Pastur G, Cellini JM, Lencinas MV. 2004. Forty years of silvicultural management in southern *Nothofagus pumilio* primary forests. *For Ecol Manage.* 201:335–347.
- Geijzendorffer IR, Roche P. 2013. Can biodiversity monitoring schemes provide indicators for ecosystem services? *Ecol Ind.* 33:148–157.
- Haines-Young R, Potschin M. 2010. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli DG, Frid C, editors. *Ecosystem ecology: A new synthesis*. Cambridge: Cambridge University Press; p. 110–139.
- Harrison PA, Berry PM, Simpson G, Haslett JR, Blicharska M, Bucur M, Dunford R, Egoh B, Garcia-Llorente M, Geamăna N, et al. 2014. Linkages between biodiversity attributes and ecosystem services: a systematic review. *Ecosyst Serv.* 9:191–203.
- Hirzel AH, Arlettaz R. 2003. Modeling habitat suitability for complex species distributions by environmental-distance geometric mean. *Environ Manage.* 32:614–623.
- Hirzel AH, Hausser J, Chessel D, Perrin N. 2002. Ecological-niche factor analysis: how to compute habitat-suitability maps without absence data? *Ecology.* 83:2027–2036.
- Hirzel AH, Hausser J, Perrin N. 2004. *Biomapper 3.1*. Division of conservation biology. Bern: University of Bern.
- Hodder KH, Newton AC, Cantarello E, Perrella L. 2014. Does landscape-scale conservation management enhance the provision of ecosystem services? *Int J Biodiv Sci, Ecosyst Ser Manage.* 10:71–83.
- Holechek JL. 1988. An approach for setting the stocking rate. *Rangelands.* 10:10–14.
- Hough RL. 2014. Biodiversity and human health: evidence for causality? *Biodiv Conserv.* 23:267–288.
- Isbell F, Calcagno V, Hector A, Connolly J, Harpole W, Reich P, Scherer-Lorenzen M, Schmid B, Tilman D, van Ruijven J, et al. 2011. High plant diversity is needed to maintain ecosystem services. *Nature.* 477:199–202.
- Ivancich H, Martínez Pastur G, Lencinas MV, Cellini JM, Peri PL. 2014. Proposals for *Nothofagus antarctica* diameter growth estimation: simple vs. global models. *J For Sci.* 60:307–317.
- Jax K, Heink U. 2015. Searching for the place of biodiversity in the ecosystem services discourse. *Biol Conserv.* 191:198–205.
- Kremen C. 2005. Managing ecosystem services: what do we need to know about their ecology? *Ecol Lett.* 8:468–479.
- Kreps G, Martínez Pastur G, Peri PL. 2012. Cambio climático en Patagonia Sur: Escenarios futuros en el manejo de los recursos naturales. Buenos Aires: Ed. INTA.
- Lencinas MV, Martínez Pastur G, Anderson C, Busso C. 2008. The value of timber quality forests for insect conservation on Tierra del Fuego Island compared to associated non-timber quality stands. *Insect Conserv.* 12:461–475.
- Lindenmayer D, Franklin JF, Löhmus A, Baker S, Bauhus J, Beese W, Brodie A, Kiehl B, Kouki J, Martínez Pastur G, et al. 2012. A major shift to the retention approach for forestry can help resolve some global forest sustainability issues. *Conserv Lett.* 5:421–431.
- Liquete C, Cid N, Lanzanova D, Grizzetti B, Reynaud A. 2016. Perspectives on the link between ecosystem services and biodiversity: the assessment of the nursery function. *Ecol Ind.* 63:249–257.
- López-Santiago CA, Oteros Rozas E, Martín-López B, Plieninger T, González E, González JA. 2014. Using visual stimuli to explore the social perceptions of ecosystem services in cultural landscapes: the case of transhumance in Mediterranean Spain. *Ecol Soc.* 19:27.
- Loreau M, de Mazancourt C. 2013. Biodiversity and ecosystem stability: a synthesis of underlying mechanisms. *Ecol Lett.* 16:106–115.
- Lucas PL, Kok M, Nilsson M, Alkemade R. 2014. Integrating biodiversity and ecosystem services in the post-2015 development agenda: goal structure, target areas and means of implementation. *Sustainability.* 6:193–216.
- Luck GW, Daily GC, Ehrlich PR. 2003. Population diversity and ecosystem services. *Trends Ecol Evol.* 18:331–336.
- Luck GW, Harrington R, Harrison PA, Kremen C, Berry PM, Bugter R, Dawson TP, de Bello F, Diaz S, Feld CK, et al. 2009. Quantifying the contribution of organisms to the provision of ecosystem services. *BioScience.* 59:223–235.
- Mace G, Norris K, Fitter A. 2012. Biodiversity and ecosystem services: a multilayered relationship. *Trends Ecol Evol.* 27:19–26.
- Martínez Pastur G, Peri PL, Cellini JM, Lencinas MV, Barrera M, Ivancich H. 2011. Canopy structure analysis for estimating forest regeneration dynamics and growth in *Nothofagus pumilio* forests. *Ann For Sci.* 68:587–594.
- Martínez Pastur G, Peri PL, Lencinas MV, García Llorente M, Martín López B. 2016a. Spatial patterns of cultural ecosystem services provision in Southern Patagonia. *Land Ecol.* 31:383–399.
- Martínez Pastur G, Peri PL, Soler R, Schindler S, Lencinas MV. 2016c. Biodiversity potential of *Nothofagus* forests in Tierra del Fuego (Argentina): tool proposal for regional conservation planning. *Biodiv Conserv.* 25:1843–1862.
- Martínez Pastur G, Soler R, Ivancich H, Lencinas MV, Bahamonde H, Peri PL. 2016b. Effectiveness of fencing and hunting to control *Lama guanicoe* browsing damage: implications for *Nothofagus pumilio* regeneration in harvested forests. *Environ Manage.* 168:165–174.
- Maskell L, Crowe A, Dunbar M, Emmett B, Henrys P, Keith A, Norton L, Scholefield P, Clark D, Simpson I, Smart S. 2013. Exploring the ecological constraints to multiple ecosystem service delivery and biodiversity. *J Appl Ecol.* 50:561–571.
- McAdam JH, Burgess P, Graves A, Rigueiro-Rodríguez A, Mosquera-Losada M. 2009. Classifications and functions of agroforestry systems in Europe. In: Rigueiro-Rodríguez A, McAdam J, Mosquera-Losada M, editors. *Agroforestry in Europe*. Amsterdam: Springer; p. 21–41.
- McCouch S, Baute G, Bradeen J, Bramel P, Bretting P, Buckler E, Burke J, Charest D, Cloutier S, Cole G, et al. 2013. Agriculture: feeding the future. *Nature.* 499:23–24.
- McCune B, Mefford MJ. 1999. *Multivariate analysis of ecological data*, Version 4.0. Gleneden Beach (OR): MjM software.
- McShane TO, Hirsch PD, Trung TC, Songorwa AN, Kinzig A, Monteferri B, Mutekanga D, Van Thang H, Dammert J, Pulgar-Vidal M, et al. 2011. Hard choices: making trade-offs between biodiversity conservation and human well-being. *Biol Conserv.* 144:966–972.
- Meli P, Rey Benayas JM, Balvanera P, Martínez Ramos M. 2014. Restoration enhances wetland biodiversity and

- ecosystem service supply, but results are context-dependent: a meta-analysis. *PLoS One*. 9:e93507.
- [MEA] Millennium Ecosystem Assessment. 2005. Ecosystems and human wellbeing: current state and trends. Washington (DC): Island Press.
- Newton AC, Hodder K, Cantarello E, Perrella L, Birch JC, Robins J, Douglas S, Moody C, Cordingley J. 2012. Cost-benefit analysis of ecological networks assessed through spatial analysis of ecosystem services. *J Appl Ecol*. 49:571–580.
- Peri PL. 2009. Relevamiento de los bosques nativos de ñire (*Nothofagus antarctica*) de Tierra del Fuego. Buenos Aires: Ed. INTA.
- Peri PL. 2011. Carbon storage in cold temperate ecosystems in Southern Patagonia, Argentina. In: Atazadeh I, editor. Biomass and remote sensing of biomass. Rijeka: Tech Publisher; p. 213–226.
- Peri PL, Bahamonde H, Lencinas MV, Gargaglione V, Soler R, Ormaechea S, Martínez Pastur G. 2016. A review of silvopastoral systems in native forests of *Nothofagus antarctica* in southern Patagonia, Argentina. *Agroforestry Systems*. doi:10.1007/s10457-016-9890-6
- Peri PL, Gargaglione V, Martínez Pastur G, Lencinas MV. 2010. Carbon accumulation along a stand development sequence of *Nothofagus antarctica* forests across a gradient in site quality in Southern Patagonia. *For Ecol Manage*. 260:229–237.
- Poirazidis K, Schindler S, Kati V, Martinis A, Kalivas D, Kasimiadis D, Wrba T, Papageorgiou AC. 2011. Conservation of biodiversity in managed forests: developing an adaptive decision support system. In: Li C, Laforteza R, Chen J, editors. Landscape ecology and forest management: challenges and solutions in a changing globe. New York: Higher Education Press, Springer; p. 380–399.
- Quijas S, Schmid B, Balvanera P. 2010. Plant diversity enhances provision of ecosystem services: a new synthesis. *Basic Appl Ecol*. 11:582–593.
- Reyers B, Biggs R, Cumming GS, Elmqvist T, Hejnowicz AP, Polasky S. 2013. Getting the measure of ecosystem services: a social-ecological approach. *Front Ecol Environ*. 11:268–273.
- Richards DR, Friess DA. 2015. A rapid indicator of cultural ecosystem service usage at a fine spatial scale: content analysis of social media photographs. *Ecol Ind*. 53:187–195.
- Running SW, Nemani RR, Heinsch FA, Zhao MS, Reeves M, Hashimoto H. 2004. A continuous satellite-derived measure of global terrestrial primary production. *Bioscience*. 54:547–560.
- Sandifer PA, Sutton-Grier A, Ward B. 2015. Exploring connections among nature, biodiversity, ecosystem services, and human health and well-being: opportunities to enhance health and biodiversity conservation. *Ecosyst Ser*. 12:1–15.
- Schindler S, O'Neill FH, Biró M, Damm C, Gasso V, Kanka R, van der Sluis T, Lauwaars S, Krug A, Pusch M, et al. 2016. Multifunctional floodplain management and biodiversity effects: A knowledge synthesis for six European countries. *Biodiv Conserv*. 25:1349–1382.
- Schindler S, Sebesvari Z, Damm C, Euller K, Mauerhofer V, Hermann A, Biró M, Essl F, Kanka R, Lauwaars SG, et al. 2014. Multifunctionality of floodplain landscapes: relating management options to ecosystem services. *Land Ecol*. 29:229–244.
- Schneiders A, Van Daele T, Van Landuyt W, van Reeth W. 2012. Biodiversity and ecosystem services: complementary approaches for ecosystem management? *Ecol Ind*. 21:123–133.
- Smith J, Pearce B, Wolfe M. 2013. Reconciling productivity with protection of the environment: is temperate agroforestry the answer? *Renew Agric Food Syst*. 28:80–92.
- Soliveres S, Manning P, Prati D, Gossner M, Alt F, Arndt H, Baumgartner V, Binkenstein J, Birkhofer K, Blaser S, et al. 2016. Locally rare species influence grassland ecosystem multifunctionality. *Phil Trans R Soc B*. 371:20150269.
- Tallis H, Kareiva P, Marvier M, Chang A. 2008. An ecosystem services framework to support both practical conservation and economic development. *Proc Natl Acad Sci*. 105:9457–9464.
- Tilman D, Fargione J, Wolff B, D'Antonio C, Dobson A, Howarth R, Schindler D, Schlesinger WH, Simberloff D, Swackhamer D. 2001. Forecasting agriculturally driven global environmental change. *Science*. 292:281–284.
- Torres AD, Cellini JM, Lencinas MV, Barrera MD, Soler R, Díaz-Delgado R, Martínez Pastur G. 2015. Seed production and recruitment in primary and harvested *Nothofagus pumilio* forests: influence of regional climate and years after cuttings. *For Syst*. 24:e-016.
- Trabucchi M, O'Farrell PJ, Notivol E, Comín FA. 2014. Mapping ecological processes and ecosystem services for prioritizing restoration efforts in a semiarid Mediterranean River Basin. *Environ Manage*. 53:1132–1145.
- Tscharntke T, Tylianakis JM, Rand TA, Didham RK, Fahrig L, Batary P, Bengtsson J, Clough Y, Crist TO, Dormann CF, et al. 2012. Landscape moderation of biodiversity patterns and processes: eight hypotheses. *Biol Rev*. 87:661–685.
- Zhao M, Running SW. 2010. Drought-induced reduction in global terrestrial net primary production from 2000 through 2009. *Science*. 329:940–943.