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Article

Effectiveness of Protected Areas in the conservation of *Nothofagus antarctica* Forests in Santa Cruz, Argentina

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Abstract

Protected areas (PA) constitute a fundamental strategy for mitigating biodiversity loss. Land-sparing approach has expanded in response to international agreements, but expansion of PA does not guarantee conservation objectives. The objective was to assess PA effectiveness in conserving *Nothofagus antarctica* forests in Santa Cruz (Argentina) evaluating human impacts (fire, animal use, harvesting). The research was conducted within pure native forests in Santa Cruz, Argentina. This province encompasses 52 protected areas, representing the highest concentration of conservation units within the forested landscapes of the country. At least eight of these areas include *N. antarctica* forests. Three land tenure categories were evaluated: protected areas (PA), buffer of 15-km from PA boundaries on private lands (BL), and private lands (PL). 103 sampling plots were established, where 38 variables were assessed (impacts, soil, forest structure, understory, animal use). Three indices were developed to analyze ecosystem integrity: forest structure (FI), soil (SI), and animal use (AI). PA presents highest FI (0.64 for PA, 0.44 for BL, 0.30 for PL) and AI (0.60 for PA, 0.55 for BL, 0.52 for PL), and together with buffer zones, the highest SI (0.43 for PA, 0.47 for BL, 0.32 for PL). PA showed superior integrity regarding compared to BL and PL, indicating effective preservation despite anthropogenic impacts.

Keywords: biodiversity; ecosystem integrity; land-sparing; monitoring; resilience

1. Introduction

Global Protected Area (PA) coverage currently encompasses 17.6% of terrestrial and inland water systems and 8.4% of marine and coastal ecosystems [1]. The Kunming-Montreal Global Biodiversity Framework sets the target of expanding the global PA network to 30% by 2030 [2,3]. Although the establishment of PA remains a central instrument for biodiversity protection and conservation, biodiversity loss continues [4,5]. Consequently, evaluating PA effectiveness is essential [6], and is typically defined as the integration of three components: PA location, management performance, and the capacity to implement the designated Management and Conservation Plan [7]. These analyses and evaluations often rely on subjective and, in some cases, self-assessed questionnaires, even though their methodological rigor has improved over time [8]. Within this context, monitoring must determine the extent to which the conservation of a PA natural and cultural

values is being achieved by assessing management effectiveness [9]. Such monitoring can be conducted from biological and ecological perspectives to evaluate the condition of ecosystems and their biodiversity [10,11]. Some studies, such as Delgado et al. [12] and DellaSala et al. [13], employ indices based on deforestation and changes in land cover and land use, although these indicators do not always reflect reductions in ecological integrity. Moreover, not all forms of degradation involve land-use changes detectable through remote sensing, yet they can still affect ecosystem integrity within PA [12,14]. Therefore, monitoring biological and ecological attributes is necessary to enable a locally grounded assessment [15].

In Argentina, only one quarter of the research conducted within National Parks (NP) aligns with the research priorities defined in their management plans. Moreover, for several NP, management plans are either unavailable or not publicly accessible [16]. With 52 PA, including seven NP, Santa Cruz is the Argentine province with the highest number of protected areas [17]. These PA protect a wide range of aquatic and terrestrial environments, including steppes and forests [18]. Notably, pure forests of *Nothofagus antarctica* (G. Forst.) Oerst. occur under different conservation statuses: in at least eight PA, three within NP (Parque Nacional Los Glaciares, Parque Nacional Patagonia, Parque Nacional Perito Moreno) and five within Provincial Reserves (Reserva Provincial Lago del Desierto, Reserva Provincial Tucu-Tucu, Reserva Provincial Península Magallanes, Reserva Provincial Punta Gruesa, Reserva Provincial San Lorenzo) as well as on private lands [19]. They occupy 1,699 km² in Santa Cruz and constitute the smallest protected area category (16% of the total *N. antarctica* forests), where important economic activities predominate [18].

Nothofagus antarctica is one of the most phenotypically plastic tree species in Andean-Patagonian forests, capable of adapting to a wide range of environmental conditions [20,21]. In addition to this plasticity, monospecific *N. antarctica* forests exhibit distinct phases of the natural life cycle (initial growth, final growth, mature, and decay), which allow for the classification of stands as even-aged (>70% of basal area represented by a single growth phase), two-aged (≥70% represented by two phases), or uneven-aged (three or more phases representing >70% of basal area) [22]. Consequently, a single conservation strategy is not feasible for this heterogeneous forest [23,24], particularly considering that it also provides important cultural, regulating, and provisioning ecosystem services [25]. In Santa Cruz province, the use of *N. antarctica* forests is primarily associated with silvopastoral systems involving domestic livestock (cattle, horses, and sheep), while a smaller proportion undergoes forestry interventions, mainly thinning, to produce posts, poles, and firewood [26]. Overgrazing, land-use change, and forestry interventions exert pressure on *N. antarctica* forests, compromising their conservation and long-term persistence [27]. Rosas et al. [18] identify *N. antarctica* forests as those presenting the greatest potential trade-offs between production and conservation of *Nothofagus* in Santa Cruz, with particular emphasis on areas located in the ecotone, where they border pastures and become important for silvopastoral production, as these environments provide both forage and shelter for domestic livestock. This creates the production-conservation dilemma, often framed by the concepts of land-sharing and land-sparing [28,29]. The land-sharing approach promotes agricultural production and biodiversity conservation within the same area and is associated with complex landscape structures in which natural habitats coexist with low-intensity production systems that employ biodiversity-friendly practices [30]. In contrast, land-sparing aims to secure the largest possible extent of contiguous, intact natural habitats, spatially separating areas designated for biodiversity protection from those allocated to production [31]. Furthermore, the sensitivity of forests to climate variability accentuates the complexity of the problem, increasing ecosystem vulnerability [32], both due to climate change itself and to its effects on meteorological variables associated with the heightened probability and occurrence of large-scale forest fires [33]. For this reason, it is important to consider resilience defined within the conceptual framework of Peri et al. [23] as the ability to maintain and/or recover ecosystem identity in the face of disturbances, as well as the species vulnerability to climatic factors. Some studies, for example, demonstrate the high resilience of *N. antarctica* to fires [34] and to harvesting, owing the resprouting capacity [23]. Determining the effectiveness of PA makes it possible to evaluate whether the proposed conservation

objectives are being met, whether current management and conservation strategies are effective, and whether alternative strategies need to be considered [35]. Therefore, the objective of this study is to determine the effectiveness of PA in conserving *N. antarctica* forests in Santa Cruz province, assessing the impact of fires, animal use, and harvesting. To address the following questions: (i) Do livestock occupancy, forest structure, and soil properties vary according to land tenure status? (ii) Does the intensity of impacts increase as we move away from PA? we propose the following hypotheses: (i) Unprotected areas exhibit greater livestock, logging, and fire pressure, which gradually decreases toward PA; (ii) Land tenure status influences forest structure (e.g., dominant height, crown cover, basal area, total over-bark volume, percentage of mature basal area, vigor, regeneration cover, seedling density and height, sapling density and height), with more conserved structures expected in PA than in unprotected areas; (iii) Private tenure status leads to negative impacts on soil (e.g., compaction, acidification, carbon loss, and reduced water retention capacity) where more intensive use occurs; and (iv) PA experience lower herbivory pressure on the understory.

2. Materials and Methods

2.1. Study Area

The study area comprises the pure *N. antarctica* forests of Santa Cruz province, Argentina. These forests exhibit different structural conditions (even-aged or uneven-aged), depending on natural disturbances (e.g., windstorms) and anthropogenic impacts over the last century (e.g., wildfires, logging, pasture establishment, livestock grazing) [18,19]. A total of 103 stands (Figure 1), each at least 2 ha in size and characterized by homogeneous forest structure, were sampled, representing the environmental heterogeneity present within Protected Areas (National Parks and Provincial Reserves) and Private Areas on farms, primarily livestock operations. A 15 km buffer zone was defined from the PA boundary, which was used to classify private-area stands into private land and buffer-area categories (Figure 1).

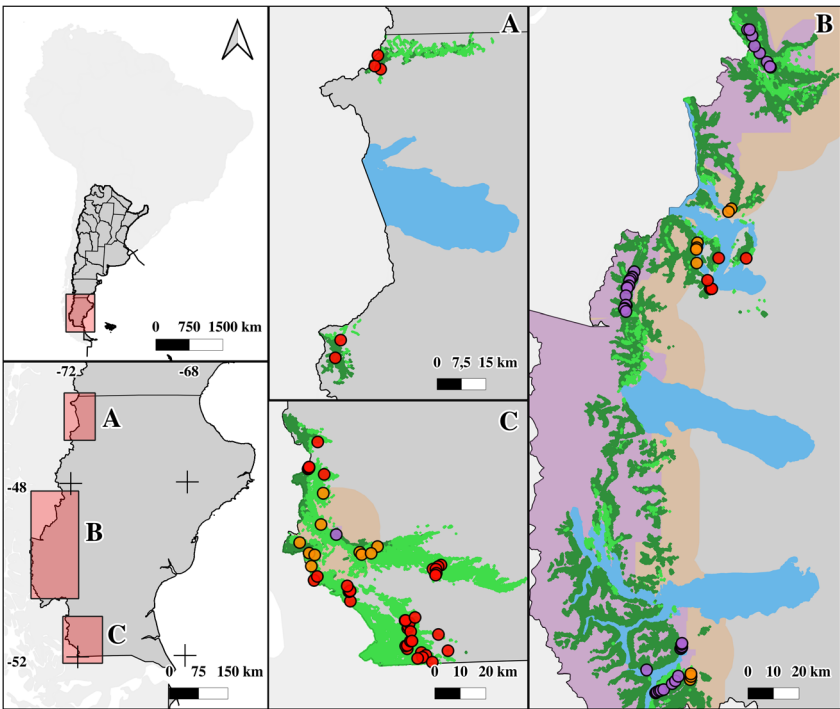


Figure 1. Map of the study area, identifying plots (dots) classified by land tenure (purple = Protected Areas, orange = buffer areas in private lands, red = private lands), and forests (light green = *Nothofagus antarctica* forests, dark green = other forest cover, violet = Protected Areas, orange = 15 km buffer, light blue = lakes).

2.2. Data Taking and Data Analysis

At each sampling unit (stand), a sampling plot was established, consisting of a 50 m transect placed in a random direction. Stands that had been harvested were identified by the presence of stumps resulting from cuts applied with different intensities (H-Int) (0 = no harvesting, 1 = light selective cutting, 2 = heavy selective cutting, 3 = clear cutting). Fire impact was recorded in stands showing substantial alterations to their original forest structure caused by fires of varying magnitude and intensity (F-Int) (1 = less than half of the canopy burned, 2 = half of the canopy burned, 3 = complete canopy mortality). For each land tenure status considered, the occurrence of harvesting (H-O), fires (F-O), and animal use (A-O) was calculated as the number of plots in which the impact was recorded relative to the total number of plots classified within that status. The presence of regeneration was recorded at every meter along the 50 m transect, and these observations were used to calculate regeneration cover (CAN, %). At the 10 m and 40 m points of the transect, forest structure was characterized using Bitterlich angle-count sampling ($K = 3-5$) [36]. For each individual recorded, the diameter at 1.3 m height (DAP), vigor (1-3, with 3 representing the highest vitality), and development phase based on bark characteristics following the classification used by Martínez Pastur et al. [22], were registered. Additionally, tree crown classes were assigned based on light capture (dominant, co-dominant, intermediate, and suppressed) [37]. Each angle-count measurement point was associated with a dominant height value (measured with a TruPulse 200 rangefinder, corresponding to the tallest tree within 50 m distance), and these were averaged to obtain dominant height per stand (DH, m). Using these data, basal area (BA, $\text{m}^2 \text{ha}^{-1}$), total over-bark volume (TOBV, $\text{m}^3 \text{ha}^{-1}$) [38], percentage of mature basal area (PBA-M), and average vigor per plot (VIG, 1-3) were calculated. Hemispheric photos (Nikon 35 mm with Sigma 8 mm lens) were taken at 10 m and 40 m of the transect to calculate canopy cover (CC, %) using Gap Light Analyzer v.2.0 software [39]. The structure (height and density) and browsing on initial regeneration (<1.3 m height) and advanced regeneration (>1.3 m height and <5 cm DBH) were surveyed in two plots, at the beginning and end of the transect, using 1 m^2 for the initial regeneration and 5 m^2 plots for the advanced regeneration. With this data, seedling density (DSE, thousand ha^{-1}), sapling density (DSP, thousand ha^{-1}), seedling height (HSE, cm), sapling height (HSP, m) and browsing damage in seedlings (BRW, %) were calculated.

To characterize the topsoil layer, we collected four soil samples along each transect using a hand soil sampler (0-30 cm depth) of known volume (200 cm^3), after removing the litterfall. Samples were weighed before and after air-drying under laboratory conditions (24°C) until a constant weight was achieved, obtaining soil water content (SWC, %) and soil bulk density (SD, g cm^{-3}), after coarse root debris and stones >2 mm were removed by sieving. The samples, sieved to 2 mm, were oven-dried at 70 °C to constant weight, and we determined: (i) soil acidity (pH) in a suspension (air-dried samples and deionized water) with a soil/water ratio of 1:2.5 [40], and (ii) total carbon (C, %) by dry combustion analysis (muffled at 500°C for 24 h) [41] and modeling [42]. Using the obtained values of effective density and total carbon percentage, the carbon content (SOC, t ha^{-1} at 30 cm depth) was calculated.

Dung counts were performed, distinguishing between herbivores, in a 4x50 m plot along the transect (200 m^2). This was used as a proxy for livestock occurrence (animals ha^{-1}), following the methodology of Martínez Pastur et al. [42], using for hares: an average defecation rate of 410 times per day, a dry matter forage requirement of 24 kg DM year^{-1} , a residual palatable biomass of 130 kg DM ha^{-1} , and a sheep equivalent (SE) of 0.075. With this data, we determined herbivore occupation, discriminating guanacos (LG, SE ha^{-1}), hares (LE, EO ha^{-1}), sheep (SHE, EO ha^{-1}), cattle (CAT, EO ha^{-1}), horses (HOR, EO ha^{-1}), domestic livestock (LIV, EO ha^{-1}), and total occupation (TSD, EO ha^{-1}). For the considered land tenure statuses, the occurrence of animal use (A-O) was calculated as the number of plots in which this impact was recorded out of the total plots classified within that status. For the occurrence of animal use, values up to 0.5 sheep equivalent were considered as zero. The impact of animal use (A-Int) was calculated by averaging domestic livestock loads.

An index evaluating the overall diversity of impacts (IMP) on forests was constructed by averaging the intensity of harvesting, fire, and livestock impacts. Samples of understory aerial biomass were collected in plots (0.25 m²) associated with the transect, cutting vegetation at ground level and ≤1.3 m. They were manually classified into palatable and non-palatable species after oven-drying at 70 °C to constant weight and then weighed to obtain the palatable biomass. Additionally, average understory height was measured. Using the palatable biomass value, the potential occupation (POT) in sheep equivalents per hectare was calculated. Three additional indices were constructed to group the analyzed variables by topic (forest, soil, animal use) and were then used to compare conservation status. Values were standardized between 0 and 1 and averaged within each index to obtain a value per plot. Variables considered positive were kept in their original form, while negative variables were inverted by subtracting them from 1. These values were then averaged by status for the final outputs. The forest index (FI) includes dominant height, canopy cover, live basal area, total over-bark volume, percentage of mature basal area, vigor, regeneration cover, seedling density, seedling height, sapling density, and sapling height, with all these variables considered positive. The soil index (SI) included soil water content, soil carbon percentage, and carbon content per hectare in the first 30 cm of depth, all considered positive, and soil bulk density, considered negative. Finally, the animal use index (AI) included direct animal load measurement variables and indirect measurements based on understory and biomass, including understory height, live dry weight, palatable dry weight, potential SE, and guanaco sheep equivalents (SE), all considered positive, and browsing percentage, hare SE, sheep SE, cattle SE, horse SE, domestic livestock SE, and total SE, all considered negative. Values close to 1 in the FI indicate forests with the highest structural attributes. For the SI, higher values represent soils with the best qualities for tree development, while higher AI values indicate greater ecosystem integrity regarding animal use intensity, e.g., higher AI corresponds to lower domestic animal use. Based on the premise that a PA is effective when it is managed to conserve ecosystem integrity across territories and ecosystem types [43], ecosystem integrity was evaluated using the generated indices, with values of 1 representing the highest levels of conservation for forest structure, soil, and animal use.

2.3. Statistical Analysis

The data were analyzed by one-way analysis of variance (ANOVA) and pairwise mean comparisons using Tukey's test, with the Statgraphics program [44]. Thirtyfive ANOVAs were performed (one for each variable: H-Int, F-Int, A-Int, IMP, DH, CC, BA, TOBV, PBA-M, VIG, CAN, DSE, HSE, DSP, HSP, SD, SWC, C, SOC, pH, UH, UAB, UPB, BRW, POT, TSD, LG, LE, SHE, CAT, HOR, LIV, AI), considering land tenure status (PL, BL, AP) as the factors.

3. Results

The occurrence and intensity of fire, harvesting, and livestock impacts varied according to conservation status (Table 1). Private lands showed the highest values for fire occurrence (FO) and fire intensity (F-Int), and the lowest values for harvesting occurrence and harvesting intensity. Private buffer lands (BL) presented the highest values for harvest occurrence and intensity (HO, H-Int) and animal use (A-O and A-Int). However, it shows fire occurrence and fire intensity values like those of protected areas, and lower than those recorded in private lands. In the status categories where the highest values of impact occurrence were recorded, the highest impact intensities were also observed. PA had the lowest impact values (IMP), while private buffer lands showed the highest. Regarding intensities, compensatory patterns were observed. For example, PL areas with higher fire intensity showed lower harvesting and animal use intensities; BL areas had higher animal use and harvesting intensity but lower fire intensity; and PA recorded intermediate values for harvesting and animal use intensities but lower fire intensities. No significant differences were found for the overall impact index.

Table 1. Mean values of occurrence and intensity of harvesting, fire and animal use for each conservation status (PL = private lands, BL = buffer areas in private lands, PA = protected areas) in *Nothofagus antarctica* forests of Santa Cruz Province, Argentina. There are also included the statistics (F = Fisher test, p = probability at <0.05) of the analyses of variance (ANOVA) for impact intensity.

Level	H-O	H-Int	F-O	F-Int	A-O	A-Int	IMP
PL	0.13	0.17	0.51	0.81	0.66	0.38	0.45
BL	0.33	0.44	0.33	0.50	0.78	0.64	0.53
PA	0.29	0.37	0.34	0.50	0.63	0.46	0.44
F		1.97		1.62		0.92	0.22
(p)		(0.145)		(0.204)		(0.402)	(0.801)

PL = private lands, BL = buffer areas in private lands, PA = protected areas. H-O = harvesting occurrence (%), H-Int = harvesting intensity (0–3), F-O = fire occurrence (%), F-Int = fire intensity (0–3), A-O = Animal use occurrence (%), A-Int = Animal use intensity (0–3), IMP = impact (0–1).

Regarding forest structure, forests in PL exhibited the lowest values for DH, CC, BA, TOBV, VIG, and HSP (Table 2), contrasting significantly with PA, which displayed the highest structural values. Seedling density showed a significant inverse response, with the highest value in PL and the lowest in PA. The distribution of development phases was analyzed using the PBA–M variable, considering the sum of aging and decaying trees. No significant differences were found among forest statues, although in all cases more than 75% of the basal area was represented by these two phases. Similarly, *N. antarctica* regeneration cover, seedling height, or sapling density showed no variation among statuses. The forest index (FI) showed significant differences, distinguishing three groups in the order PA > BL > PL. Although protected areas showed high values for H-O and H-Int, they also exhibited high values for forest structure variables (DH, CC, BA, TOBV). Forests in buffer zones, which reported harvesting values like those in protected areas, displayed a reduction in their FI. In contrast, forests on private lands, which had low FI values, also showed lower harvesting impacts.

Table 2. Mean values of forest structure and regeneration variables for different levels of conservation status (PL = private lands, BL = buffer areas in private lands, PA = protected areas) estimated in *Nothofagus antarctica* forests of Santa Cruz Province, Argentina. There are also statistics included (F = Fisher test, p = probability at <0.05) of the analyses of variance and the pairwise mean comparison.

Level	DH	CC	BA	TOBV	PBA–M	VIG
PL	5.2a	45.48a	17.7a	64.9a	89.35	1.78a
BL	7.9b	65.71b	24.9ab	110.7a	80.58	1.95a
PA	9.1b	76.51b	37.9b	177.2b	89.35	2.35b
F	27.48	20.98	12.46	18.92	0.98	22.69
(p)	(<0.001)	(<0.001)	(<0.001)	(<0.001)	(0.378)	(<0.001)
Level	CAN	DSE	HSE	DSP	HSP	FI
PL	9.4	48.8b	21.8	1.53	1.78a	0.30a
BL	4.3	17.5ab	29.7	1.24	1.94ab	0.44b
PA	3.3	8.1a	21.0	1.56	2.48b	0.64c
F	2.49	4.29	1.53	0.05	7.58	27.54
(p)	(0.088)	(0.016)	(0.221)	(0.955)	(0.001)	(<0.001)

DH = dominant height (m), CC = crown cover (%), BA = basal area (m² ha⁻¹), TOBV = total over bark volume (m³ ha⁻¹), PBA–M = percentage of basal area mature (%), VIG = vigor (1–3), CAN = regeneration cover (%), DSE = seedling density (thousand ha⁻¹), HSE = seedling height (cm), DSP = sapling density (thousand ha⁻¹), HSP = sapling height (m), FI = forest index (0–1). Different letters in the same column indicate significant differences by Tukey test (p <0.05).

Soil variables (Table 3) showed significant differences, except for pH. Higher soil bulk densities were found in PL, while the lowest values were observed in PA. SWC, C, and SOC presented lower

values in PL, and higher values in PA for SWC and in BL for C and SOC. The soil index (SI) showed significant differences, following the order PA = BL > PL. Private lands were the most affected by fire occurrence and intensity, which may explain the lower SI values. Specifically, private lands with the lowest FI values coincide with the lowest SI values. However, this trend was less distinct in buffer zones and protected areas; while both fell into the same SI group with the highest values, they differed in terms of FI, with protected areas exhibiting higher values. The areas most affected by fire occurrence and intensity (PL) are those that exhibit the lowest SI values.

Table 3. Mean values of soil variables for different levels of conservation status (PL = private lands, BL = buffer areas in private lands, PA = protected areas) measured in *Nothofagus antarctica* forests of Santa Cruz Province, Argentina. There are also statistics included (F = Fisher test, p = probability at <0.05) of the analyses of variance and the pairwise mean comparison.

Level	SD	SWC	C	SOC	pH	SI
PL	0.905b	29.5a	5.47a	13.2a	5.2	0.32a
BL	0.775ab	37.6ab	9.82b	18.0b	5.4	0.47b
PA	0.723a	67.9b	7.82b	14.9ab	5.3	0.43b
F	8.58	5.19	6.39	5.52	0.68	7.05
(p)	(<0.001)	(0.007)	(0.003)	(0.005)	(0.507)	(0.001)

SD = soil bulk density (gr cm⁻³), SWC = soil water content (%), C = total soil carbon (%), SOC = soil organic carbon (tn ha⁻¹ 30 cm), SI = soil index (0–1). ANOVA results were presented for impact intensity. Different letters in the same column indicate significant differences by Tukey test (p <0.05).

For direct and indirect animal use variables (Table 4), significant differences were found for UH and SHE. For UH, the highest values were observed in PA and the lowest in PL, while for SHE, the lowest values were found in PA and the highest in PL. Palatable biomass (UPB) and POT showed marginally significant differences, being higher in PA and lower in PL for both. No significant differences were found for UAB, BRW, TSD, LG, LE, CAT, HOR, or LIV. The animal use index (AI) showed significant differences, distinguishing two groups and one with intermediate values, following the order PA > BL > PL; this corresponds to lower domestic animal use in protected areas. This finding aligns with the A–O value, which, despite being the lowest, was like that found in unprotected buffer zones, as well as with the intermediate A–Int value. Although only SHE showed significant differences among the direct animal use variables (TSD, LG, LE, SHE, CAT, HOR, LIV), browsing values (BRW) on regeneration were lower in PA. In turn, this could explain the greater heights of HSP and UH observed in PA. Conversely, in areas with higher animal presence (PL), indicated by the lowest AI, we observed poorer soil conditions, as reflected by the SI.

Table 4. Mean values of understory vegetation, biomass and herbivory variables according to different levels of conservation status (PL = private lands, BL = buffer areas in private lands, PA = protected areas) measured in *Nothofagus antarctica* forests of Santa Cruz Province, Argentina. There are also included the statistics (F = Fisher test, p = probability at <0.05) of the analyses of variance for impact intensity, and the pairwise mean comparison.

Level	SD	SWC	C	SOC	pH	SI
PL	0.905b	29.5a	5.47a	13.2a	5.2	0.32a
BL	0.775ab	37.6ab	9.82b	18.0b	5.4	0.47b
PA	0.723a	67.9b	7.82b	14.9ab	5.3	0.43b
F	8.58	5.19	6.39	5.52	0.68	7.05
(p)	(<0.001)	(0.007)	(0.003)	(0.005)	(0.507)	(0.001)

Indirect variables: UH = understory height (m), UAB = understory alive plant dry biomass (kg ha⁻¹), UPB = palatable UAB (kg ha⁻¹), BRW = browsing damage seedling (%), POT = potential stocking density based on food availability (sheep equivalent per hectare, SE ha⁻¹). Direct variables: TSD = total stocking density (SE ha⁻¹), LG = Lama guanicoe stocking density in sheep equivalents (SE ha⁻¹), LE = *Lepus europaeus* stocking density (SE ha⁻¹), SHE = sheep stocking density (SE ha⁻¹), CAT = cattle stocking density (SE ha⁻¹), HOR = horses stocking density

(SE ha⁻¹), LIV = livestock (cattle, sheep, horses) stocking density (SE ha⁻¹). AI = animal use index (0–1). Different letters in the same column indicate significant differences by Tukey test (p <0.05).

The generated indices allowed us to examine relationships among soil properties, forest structure, and animal use. In all cases, BL exhibited the highest heterogeneity, with greater variability than PL and PA, positioning it as an intermediate condition between the two (Figure 2). Considering BL forests as intermediate between the protection levels of PA and PL, PA consistently showed the highest FI and AI values (Figure 2A). Analysis of the soil index revealed two distinct groups: PL on one hand, and BL together with PA on the other; however, along the FI axis, three groups emerged clearly, with buffer zones occupying an intermediate position (Figure 2B). The animal use index highlighted similarities between private lands and buffer zones, as well as between buffer zones and protected areas, while also distinguishing PL from PA. When considered alongside the soil index, areas with higher integrity in terms of animal use (PA and BL) also exhibited superior soil characteristics, reflected in higher SI values (Figure 2C).

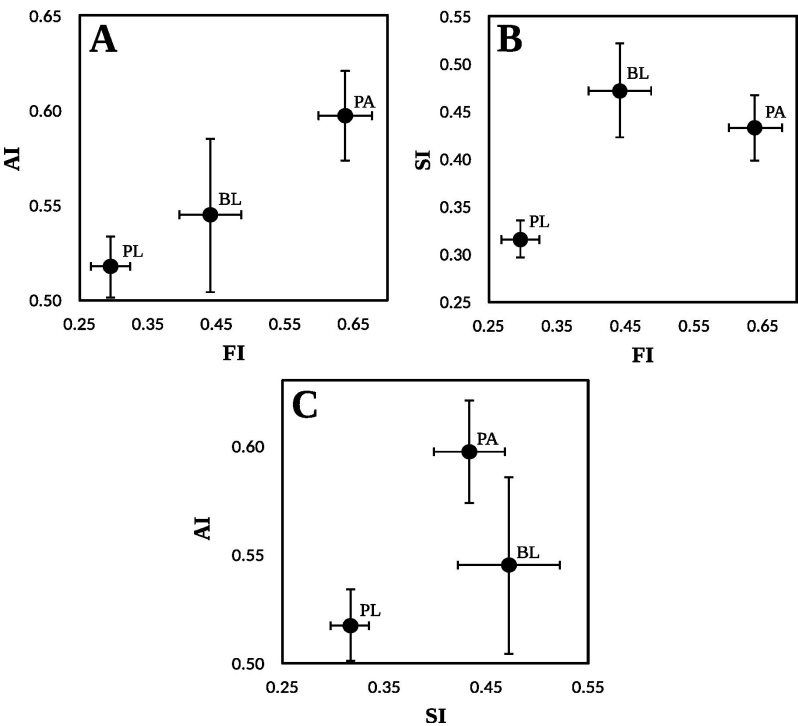


Figure 2. Relationships between indices (FI = forest index in Table 2), SI = soil index in Table 3, AI = animal use index in Table 4) calculated for *Nothofagus antarctica* forests of Santa Cruz Province, Argentina: (A) forest structure vs. animal use, (B) forest structure vs. soil, (C) soil vs. animal use. The bars represent the standard error for both axes.

4. Discussion

Although PA benefit from formal protection status, impacts from harvesting, animal use, and fire were observed within the sampled areas, consistent with studies describing the occurrence and management challenges of fires within PA in Chile [45]. The fires were intentional, aimed at opening surfaces for pastures to provide grazing for livestock [46]. Fires linked to agricultural activities are also a significant issue in other regions, such as Serbia, where they affect PA, including the Special Nature Reserve Carska Bara [47]. Added to this is animal browsing, which limits efficient regeneration due to high animal stocking rates [48]. Faúndez Pinilla et al. [45] report fires that affected PA but originated outside their administrative boundaries, highlighting the importance of considering activities within the buffer zone [49,50]. Such vulnerability at the boundaries is consistent with Sarkar et al. [51], who argue that park edges are prone to resource extraction. Therefore,

establishing large PA becomes crucial to minimizing these edge effects, as they provide a better surface-to-volume ratio [52]. This strategy facilitates the preservation of pristine core areas, consistent with observations in broad-leaved, coniferous, and mixed forests in China [53], where optimal wildlife habitats were found exclusively in the deep interior of the park. The results of this study show that the highest intensity and occurrence of forest harvesting and livestock use are found in the delimited buffer area [54]. This may reflect the effectiveness of PA in limiting these activities within their boundaries [55], though not completely, highlighting the need for increased efforts to exclude such impacts [56]. Likewise, attention to buffer areas is important, as the effects of these activities can be displaced toward protected areas, such as through fires or livestock movements [57]. Similarly, Sarkar et al. [51] documented various illicit activities (timber extraction for firewood or fencing, livestock grazing, wildlife hunting, and the harvesting of plants and wetland reeds for traditional use) occurring within Kibale National Park in Uganda. Crucially, it must be acknowledged that many of these productive practices predate the establishment of protected areas and their subsequent implementation, monitoring, and effectiveness evaluation [56]. These activities leave a lasting ecological footprint, necessitating long-term efforts to reverse their effects and ensure the efficacy of exclusion measures [53]. This indicates that the presence of domestic livestock has constituted a persistent challenge for PA in Argentina for over three decades [58,59]. However, this is a global conservation issue, also observed in reserves such as the Giant Panda National Park. As noted by Li et al. [53], the persistence of livestock is attributed to the significant effort required for exclusion, which involves both logistical constraints and potential territorial conflicts among stakeholders. Given that forests characterized by lower harvesting occurrence and intensity are located within productive private lands, the necessity of integrating protection measures into land-sharing landscapes becomes evident [60,61]. Furthermore, *N. antarctica* forests within PA exhibit the highest forest structure values, despite the impacts described. This demonstrates the resilience of these ecosystems and their capacity for structural recovery following disturbance [62,63]. Conversely, this analysis is influenced by site quality gradients; specifically, superior site conditions are found near the mountain range, declining towards the provincial interior within the ecotone [64]. This indicates that the sites selected for PA, while demonstrating resilience in meeting conservation objectives, have primarily preserved *N. antarctica* forests characterized by the highest forest structure values, such as dominant height, canopy cover, total volume, and basal area [65]. The highest initial *N. antarctica* regeneration densities were recorded on private lands. This trend may be attributed to lower stand density (lower canopy cover), which enhances light availability and minimizes competition for soil resources [66,67]. Consequently, the lower regeneration values observed in PA, where canopy cover is higher, are consistent with these ecological dynamics [68]. This finding further highlights the resilience of *N. antarctica* forests, including those located in sites of lower quality and those subjected to disturbance. Conversely, significant differences were observed in sapling heights, likely attributable to greater animal use intensity as evidenced by higher browsing (BRW) values [69,70]. Regarding soil parameters, forests conserved within PA, along with those in buffer zones, were associated with the highest Soil Index values, characterizing soils with lower bulk density, higher water content, and mean SOC and C levels [42]. This trend may be attributed to the greater exposure of unprotected lands to disturbances, as well as the land-use history in areas where forests were cleared for livestock grazing [34,71]. The observed SOC levels partially align with the distribution described by Martínez Pastur et al. [72], who reported an increasing carbon content gradient from east to west across Patagonia. Consistent with their analysis, the peak SOC values found in the buffer zone may be explained by topographic or climatic drivers, wherein lower temperatures, higher precipitation, and higher elevations favor SOC accumulation. Water content is higher in PA, a pattern likely attributed to canopy cover as observed by Koelemeijer et al. [73], who noted that cover levels characteristic of primary forests exhibits the highest percentages of soil moisture and water storage [74]. This suggests that forest structures within PA are better preserved. Similarly, while sites located in the mountain range receive higher precipitation [75], this input is also intercepted by the tree canopy [76]. In the *Nothofagus* forests of Santa Cruz, ecosystem services and biodiversity are

positively correlated with forest cover [18]. Consequently, the observed increase in cover from private lands to PA suggests that PA are situated in locations where the forest possesses the greatest potential to harbor biodiversity and provide ecosystem services [77]. Furthermore, according to Rosas et al. [18], *N. antarctica* forests exhibit a significant prevalence of potential trade-off areas, indicating that this species is crucial for both provisioning ecosystem services and biodiversity conservation. This duality raises significant challenges for forest protection [78,79], as impacts from harvesting and livestock use persist even within PA. Given that ecotonal *N. antarctica* forests are situated closer to the steppe than to the cordillera, these authors emphasize their importance for livestock production, specifically through the combined provision of forage and shelter [80]. Consequently, these environments warrant special attention as potential sites for future conservation initiatives or the establishment of new formal reserves [81]. Upon comparing the indices, specifically the animal index (AI) and the forest index (FI) (Figure 2A), it is observed that forests within buffer zones exhibit greater similarity to private lands than to PA. Based on the preceding discussion, it is determined that PA are effective in mitigating domestic animal use (specifically through the exclusion of sheep grazing), thereby enhancing ecosystem integrity [53], as well as in maintaining forest structural integrity [82]. This preservation safeguards the forest's potential to sustain both biodiversity and ecosystem services [25]. Regarding the forest index (FI) and the soil index (SI), forests within buffer zones cluster closer to PA than to private lands. These distinctions are primarily driven by understory height and sheep stocking density. The former is higher in PA and BL, whereas the latter exhibits a decreasing gradient from PL towards PA. These variables are interrelated, as animal stocking density is the principal factor negatively affecting understory height [83]. This indicates that soil conditions in buffer zones are comparable to those in PA. Furthermore, when analyzing the relationship between the animal index and the Soil Index, forests in PA and BL are observed to exhibit superior soil conditions alongside better conservation status regarding animal use. Even though soil supports ecosystems, soil conservation via protected areas is still lacking [84]. Our results provide values that enable a comparison of conditions

5. Conclusions

The effectiveness of PA in Santa Cruz was quantitatively assessed in comparison to private lands and buffer areas. Contrary to the initial hypothesis, protected areas exhibited higher harvesting impacts than private lands, while animal use intensity was comparable between the two management categories. Although fire incidence is lower, such events still occur. Nevertheless, protected areas in Santa Cruz successfully preserve the integrity of forest structure and vegetation (regarding animal use and soil properties) when compared to designated buffer zones (15 km) and private lands lacking specific biodiversity conservation legislation. Consequently, the buffer zone functions as a transitional area between unprotected private lands and protected areas. In all analyses, protected areas were clearly distinct from private lands. However, sustained actions for livestock exclusion, harvest regulation, and fire management remain necessary, as these impacts persist within protected areas. Ongoing management measures, particularly monitoring and fire control, are essential.

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Abbreviations

The following abbreviations are used in this manuscript:

PA	Protected areas
BL	Buffer lands
PL	Private lands
FI	Forest structure index
SI	Soil index
AI	Animal use index
NP	National parks

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