

Changes derived by the silvopastoral management in *Nothofagus antarctica* forests of Tierra del Fuego compared to other productive environments

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Abstract

Silvopastoral systems (SPS) are proposed as one option that promotes a balance between productive alternatives and ecological values. The objective was to determine the changes generated by SPS (thinning + livestock) in *Nothofagus antarctica* (ñire) forests compared to other productive environments (unmanaged forests, dry grasslands, wet grasslands). Changes in the main components (tree, environment, forage, animal, biodiversity) were analyzed. A total of 16 areas (4 treatments x 4 replicates) were selected in Ea. El Roble (Tierra del Fuego, Argentina), where 50 variables were surveyed through plots (forest structure, forage, animals, biodiversity) and sampling in soil pits. Indices for each component and univariate analyses were performed to compare the studied environments. The results showed that SPS generates significant changes in the studied components, causing positive and negative synergies on the studied variables. These changes promote new productive environments with intermediate characteristics between forests and grasslands (dry and humid). It was observed that tree roots, unlike herbs and grasses, are located not only in the upper layer, but also appear in higher percentages at deeper layers. These results allowed us to identify the factors of change in the studied components, as well as potential indicators for monitoring. Understanding the dynamics and interactions among the different ecosystem components, allowed to generate new proposals that must be economically viable (e.g. management costs vs. growth of trees and forage) and environmentally sustainable (e.g. conservation of the original biodiversity).

Introduction

Cattle management in Tierra del Fuego (Argentina) varies across the landscape and is related to socio-economic changes that impact over the profitability of the ranches (Martínez Pastur et al. 2016; Alonso et al. 2020; Peri et al. 2022). Ranches present highly heterogeneous ground cover, where the main economic activity is related to cattle breeding, including calf rearing and fattening for sale as steers and heifers (approx. 1 animal per 9–15 ha) (Favoretti et al. 2018). Cattle management is extensive and was conducted in large paddocks (> 500-1,000 ha) that consist of a combination of different types of productive environments, such as dry or wet grasslands, pastures, *Nothofagus antarctica* (ñire) forests under natural dynamics, and to a lesser extent, managed forests with different impact degrees and indirect anthropogenic modifications (e.g. species invasion). The impact degree is mainly related to their accessibility (Soler et al. 2022). The main challenge for cattle management is optimizing the use of the different environments throughout the year, with a strong contrast between the growing season and the long winter period. Some environments are prioritized and valued for forage productivity (e.g. wet grasslands), while others are undervalued (e.g. native forests with close canopy cover), although they play a fundamental role in providing protection and comfort to animals to face adverse weather events (Peri et al. 2017a; Favoretti et al. 2018; Martínez Pastur et al. 2022).

Historical management promoted the replacement of ñire forests with native grasslands or artificial pastures with introduced species. However, these practices required significant investments (e.g. clearing, wood waste management, and planting pastures), and often resulted in negative synergies (e.g. abundant tree regeneration that created closed secondary forests that impede cattle access or make herding difficult). Furthermore, during the last decades, rancher owners and local people have a negative perception of these high-impact practices (Gea et al. 2004; Martínez Pastur et al. 2016; Gamondes Moyano et al. 2016). More recently, silvopastoral systems (SSP) are proposed as a more friendly alternative, aiming to integrate multiple objectives (land-sharing) (Fischer et al. 2014), combining economic, conservation and provision of ecosystem service purposes (Peri et al. 2016). SSP propose to modify the balance among the forest components (e.g. trees and forage) in order to improve certain aspects (e.g. animal production) while maintaining specific levels of others (e.g. biodiversity) (Martínez Pastur et al. 2020; Bussoni et al. 2021; Peri et al. 2022). The theoretical proposals varied according to the forest types and specific site conditions (e.g. hydrological regime) (Peri et al. 2017a), being necessary the understanding of the impacts at each particular situation. In Tierra del Fuego, experiences with SSP present particular challenges, mainly related to extreme climate conditions and cattle management, e.g. existence of synergies (positives and negatives) and conflicts (cattle production vs. conservation). Besides, it is essential to define the correlation between production variables (e.g. forage supply and cattle use) and prioritize the SSP implementation within the ranches (e.g. Martínez Pastur et al. 2021, 2022).

Additionally, it is important to identify economic and degradation indicators within the national policy framework (MBGI - Integrated Forest Management with Livestock) using long-term monitoring strategies (Peri et al. 2021). Soil is an essential component of the natural ecosystems (Gargaglione et al. 2014), but often is overlooked in the monitoring programmes, and it is rarely considered into scientific studies or decision-making (Neher 1999; López Mársico et al. 2020). Soils are responsible for supporting and regulating several ecosystem services (Delgado-Baquerizo et al. 2020), and little is known about the impact or changes due to SSP. However, soils are the main factor that influence over functionality, and also are the main recipient of the impacts (Lehmann et al. 2020). In this context, adaptive management based on local results is necessary, rather than generic SSP proposals. The objective was to quantify the changes produced by SSP management in ñire forests of Tierra del Fuego (Argentina) in the different components (tree, environment, forage, animal, biodiversity) compared to other productive environments (unmanaged forests, dry and wet grasslands). We want to answer: (i) What are the differences between managed and unmanaged forests and other associated environments (dry and wet grasslands)?, (ii) What changes does SSP management (thinning + cattle use) compared to unmanaged forests?, and (iii) How does SSP management influence the main characteristics across the soil profile? These results aim to identify the main factors of change in the SSP components, as well as indicators for long-term monitoring, determining the productive advantages and functionality losses.

Materials and methods

Study site and selection of environments

Sixteen areas (4 treatments x 4 replicates) of at least 2 ha were selected within the productive landscapes of El Roble ranch (13,145 ha) in Tierra del Fuego, Argentina (54°05'37" SL, 67°40'54" WL, 68 m.a.s.l.). The economic activities are based on year-round extensive livestock breeding, with rotational stocking rates among paddocks (seasonal or annual), depending on vegetation composition, accessibility, and management objectives. The studied environments were: (i) primary unmanaged forests (PF), (ii) harvested forests (1950–1970) with partial overstory canopy removal (HF), comparable to SSP thinning proposals; (iii) dry grasslands dominated by *Festuca gracillima* and *Empetrum rubrum* (DG), and (iv) wet grasslands (meadows) dominated by *Juncus*

scheuzeroides, *Carex curta*, *C. macloviana*, and *Caltha sagittata* near rivers and streams (WG) (Fig. 1). Studied environments are under livestock (cattle) and *Lama guanicoe* (guanaco) grazing throughout the year.

Sample collection and analyses

Forest structure was characterized by two angular counts ($K = 3-5$) at the beginning and end of a 50-m transect, where trees (> 5 cm) was measured for diameter, height, social class, developmental stage, and other necessary characteristics for biometric modelling (Martínez Pastur et al. 2021). These measurements provided tree density (ind.ha^{-1} , DEN), basal area ($\text{m}^2.\text{ha}^{-1}$, BA), total over bark volume ($\text{m}^3.\text{ha}^{-1}$, TOBV), and growth ($\text{m}^3.\text{ha.yr}^{-1}$, GRO). Hemispherical photos (Nikon 35 mm and Sigma 8 mm lens) were also taken to obtain crown cover (% CC) and total radiation reaching to the ground (% TR) using Gap Light Analyzer (Frazer et al. 2001). Regeneration was measured by: (i) two 1 m² plots for initial regeneration (< 1.3 m), and (ii) two 5 m² plots for advanced regeneration (> 1.3 m and < 5 cm diameter). These data provided density of initial (thousand.ha^{-1} , DRI) and advanced regeneration (thousand.ha^{-1} , DRA). Additionally, understory species cover (50-point intersections) were recorded. Woody debris (> 1 cm), bare soil, and faeces from large herbivores were also measured, as well as understory biomass samples to calculate palatability (0.25 m²). These measurements provided plant species richness (n, RIC), native species (n, RICN), exotic species (n, RICE), cover (% COV), dicot cover (% COVD), monocot cover (% COVM), exotic species cover (% COVE), native species cover (% COVN), palatable species cover (% COVP), ratio of exotic and total cover (% EXO), ratio of productive degradation-indicator species (% DEGP), ratio of environmental degradation-indicator species (% DEGE), regeneration cover (% COVR), woody debris cover (% COVW), woody debris volume ($\text{m}^3.\text{ha}^{-1}$, VW), bare soil (% BS), dry weight forage biomass (kg.ha^{-1} , FB), alive forage biomass (kg.ha^{-1} , FBA), palatable biomass (kg.ha^{-1} , FBP), palatable dicot biomass (kg.ha^{-1} , FBD), palatable monocot biomass (kg.ha^{-1} , FBM), guanaco density (ovine equivalents OE.ha^{-1} , GUA), livestock density (OE.ha^{-1} , LIV), total animal density (OE.ha^{-1} , ANI), potential animal density based on available forage (OE.ha^{-1} , PANI), dry matter digestibility (% DMD), neutral detergent fibre (% NDF), metabolizable energy content (Mcal, MEC), and crude protein content (% CPC). Besides, composite soil samples ($n = 4$) from 30 cm profile were collected, obtaining soil density (g.cm^3 , SD), moisture (% SM), acidity (pH), carbon (% C), nitrogen (% N), and available phosphorus (ppm, P). We calculate soil content in 30 cm of organic carbon (kg.m^2 , SOC), nitrogen (kg.m^2 , SN), and phosphorus (kg.m^2 , SP). We also sampled a 50 cm deep soil pit associated to each transect (samples every 10 cm) obtained the variables described before, and carbon content of particulate organic matter (%C, POM), dissolved organic matter associated with silt and clay minerals (%C, DOM), rocks and > 2 mm sand (% soil, ROC), tree and shrub roots (% soil, RTRE), herb and grass roots (% soil, RFOR), and total roots (% soil, RTOT). Analyses and calculations were described by van Soest et al. (1991) and modifications (Martínez Pastur et al. 2020, 2021, 2022).

Statistical analyses

Variables were categorized into different components, standardized (values from 0-100), and averaged to obtain different indices: (i) tree component (I-TREE), (ii) environmental component (I-ENV), (iii) forage component (I-FOR), (iv) animal component (I-ANI), and (v) biodiversity component (I-BIO) (see Tables 1 and 2). Some variables included inverse ratios (100 - index) as they negatively influenced the studied components (SD, BS, DEGP, DEGE, GUA, NDF, RICE, COVE, EXO). Means and standard errors of indices were graphically compared among environments (PF, HF, DG, WG), and we analysed: (i) one-way analysis of variance (ANOVA) for variables and indices, considering environments as main factors, and (ii) multiple ANOVAs for soil characteristics at depths, considering environments and soil depth as main factors. Mean comparisons were performed using Tukey test ($p < 0.05$).

Results

Variations in components among the studied productive environments

The tree component index (I-TREE) showed significant differences, with $\text{PF} > \text{HF} > \text{DG-WG}$ (Table 1A). The specific components also showed significant differences and similar trends, except for DRA, which not differed among environments. The environmental component index (I-ENV) showed significant differences, where HF differed from other environments (Table 1B). Some variables increased when forest structure decreased (TR and SM), while others were directly related to harvesting. Soil density (SD) significantly increased in $\text{HF} > \text{DG-PF} > \text{WG}$, affecting nutrient contents (e.g. SP). Soil nitrogen content also increased in HF compared to PF and DG but was lower than WG, where it reached the highest values. Soil acidity decreased when tree cover diminished ($\text{PF} > \text{HF} > \text{DG}$), but increased in WG, possibly due to higher SOC, although this variable did not differ among treatments. Finally, BS was higher in dry grassland ($\text{DG} > \text{HF} > \text{PF}$) and nearly absent in WG.

Table 1

Analyses of variance for forest structure (A) and environmental (B) components for the different environments (PF = primary forests, HF = harvested forests, DG = dry grasslands, WG = wet grasslands).

(A)	CC	DEN	BA	TOBV	GRO	COVR	DRI	DRA	COVW	VW
PF	73.1c	1789.2b	34.5c	172.4c	1.9b	17.0b	205.0b	22.5	17.0b	78,5b
HF	53.7b	599.4ab	22.5b	117.8b	1.2b	9.0ab	< 0.1a	< 0.1	9.0ab	35,3ab
DG	7.1a	< 0.1a	< 0.1a	< 0.1a	< 0.1a	< 0.1a	< 0.1a	< 0.1	< 0.1a	< 0,1a
WG	6.6a	< 0.1a	< 0.1a	< 0.1a	< 0.1a	< 0.1a	< 0.1a	< 0.1	< 0.1a	< 0,1a
F	82.53	4.93	54.00	56.84	14.47	8.55	6.55	2.93	8.55	5,73
(p)	(< 0.001)	(0.018)	(< 0.001)	(< 0.001)	(< 0.001)	(< 0.001)	(0.007)	(0.077)	(0.003)	(0,011)
(B)	TR	SD	SM	SOC	SN	SP	pH	BS	I-ENV	
PF	37.2a	0.71b	22.2a	14.9	0.68a	0.0042b	5.5c	6.0a	42.1a	
HF	61.9b	0.97c	12.9a	14.3	0.73ab	0.0054c	4.8ab	7.5ab	38.5a	
DG	91.2c	0.80b	28.6a	13.9	0.69a	0.0041b	4.3a	21.5b	34.6a	
WG	97.9c	0.30a	133.8b	18.7	1.11b	0.0015a	5.0bc	< 0.1a	70.3b	
F	30.59	94.22	23.30	3.02	4.69	39.21	8.53	6.73	16.94	
(p)	(< 0.001)	(< 0.001)	(< 0.001)	(0.071)	(0.022)	(< 0.001)	(0.003)	(0.007)	(< 0.001)	

F = Fisher's test, p = probability where different letters indicate differences by Tukey test ($p < 0.05$). CC = crown cover (%), DEN = tree density (ind.ha^{-1}), BA = basal area ($\text{m}^2.\text{ha}^{-1}$), TOBV = total over bark volume ($\text{m}^3.\text{ha}^{-1}$), GRO = growth ($\text{m}^3.\text{ha}.\text{yr}^{-1}$), COVR = regeneration cover (%), DRI = initial regeneration density (thousand.ha^{-1}), DRA = advanced regeneration density (thousand.ha^{-1}), COVW = woody debris cover (%), VW = woody debris volume ($\text{m}^3.\text{ha}^{-1}$), I-TREE = tree component index, TR = total radiation reaching to the ground (%), SD = soil density (g.cm^3), SM = soil moisture (%), SOC = soil organic carbon (kg.m^2), SN = soil nitrogen content (kg.m^2), SP = soil phosphorus content (kg.m^2), pH = soil acidity, BS = bare soil (%), I-ENV = environmental component index.

The forage component index (I-FOR) showed significant differences, which were lower in forests (PF > HF) than in grasslands (WG > DG) (Table 2A). The dry weight forage biomass (FB and FBA) was higher in grasslands (DG > WG) than in forests (HF > PF). However, when considering palatable biomass (FBP), it was higher in WG than in the other environments. This palatable plant material did not vary in dicot species (FBD) but did in monocot species (FBM). The cover of palatable species was also highest in WG, followed by PF > HF > DG. Plants indicating environmental degradation (DEGE) were higher in forests (HF > PF) than in grasslands (DG > WG), while plants indicating productivity degradation (DEGP) did not showed significant differences. The animal component index (I-ANI) was lower in DG compared to the other studied environments (Table 2B). The potential animal density based on available forage (PANI) showed significant differences among environments, following the patterns described for the forage components (WG > DG-PF-HF). However, the observed density of wild and domestic animals (GUA, LIV, ANI) did not significantly differ among the studied environments. Palatability variables (DMD, NDF, MEC) showed significant differences, where DG differed from other environments and crude protein content (CPC) did not show differences.

Table 2

Analyses of variance for forage (A), animal (B) and biodiversity (C) components for the different environments (PF = primary forests, HF = harvested forests, DG = dry grasslands, WG = wet grasslands).

(A)	FB	FBA	FBP	FBD	FBM	COVP	DEGP	DEGE	I-FOR	
PF	774.2a	497.3a	453.6a	82.9	370.7a	128.4b	0.22	2.52a	26.2ab	
HF	1127.8a	714.1a	347.7a	66.2	281.5a	110.0ab	0.16	9.51b	19.3a	
DG	5736.5b	3932.8b	456.7a	3.3	453.4a	50.7a	< 0.01	0.79a	41.6b	
WG	3273.1ab	2283.5ab	2283.5b	506.7	1776.8b	168.2b	0.09	0.14a	64.6c	
F	6.33	5.05	20.87	2.53	23.10	10.47	2.02	27.71	15.85	
(p)	(0.008)	(0.017)	(< 0.001)	(0.106)	(< 0.001)	(0.001)	(0.165)	(< 0.001)	(< 0.001)	
(B)	GUA	LIV	ANI	PANI	DMD	NDF	MEC	CPC	I-ANI	
PF	0.04	1.01	1.05	0.99a	64.5b	31.2a	2.33b	7.67	48.6b	
HF	0.04	4.31	4.35	0.66a	62.9b	33.3a	2.27b	7.78	54.2b	
DG	< 0.01	1.01	1.01	1.00a	55.2a	43.1b	1.99a	5.65	20.1a	
WG	< 0.01	2.28	2.28	6.62b	62.3b	34.1a	2.24b	6.24	53.8b	
F	1.00	3.25	3.36	20.87	23.02	23.02	23.02	3.35	7.12	
(p)	(0.426)	(0.059)	(0.055)	(< 0.001)	(< 0.001)	(< 0.001)	(< 0.001)	(0.055)	(0.005)	
(C)	RIC	RICN	RICE	COV	COVD	COVM	COVE	COVN	EXO	I-BIO
PF	37.8c	31.8b	6.0b	166.5b	76.4	70.5a	33.6bc	132.9a	22.1b	50.4b
HF	36.8bc	31.0b	5.8b	251.5c	57.6	80.1a	65.6c	83.3a	49.7c	34.9a
DG	28.2ab	27.0ab	1.2a	116.3a	59.5	47.4a	0.6a	115.8a	< 0.1a	51.0b
WG	24.2a	20.2a	4.0b	149.0ab	67.9	141.1b	15.8ab	235.8b	11.5ab	60.9b
F	10.41	7.76	14.84	25.68	0.90	10.44	12.92	30.91	18.33	11.86
(p)	(0.001)	(0.004)	(< 0.001)	(< 0.001)	(0.471)	(0.001)	(< 0.001)	(< 0.001)	(< 0.001)	(< 0.001)

F = Fisher's test, p = probability where different letters indicate differences by Tukey test ($p < 0.05$). FB = dry weight forage biomass ($\text{kg} \cdot \text{ha}^{-1}$), FBA = alive forage biomass ($\text{kg} \cdot \text{ha}^{-1}$), FBP = palatable biomass ($\text{kg} \cdot \text{ha}^{-1}$), FBD = palatable dicot biomass ($\text{kg} \cdot \text{ha}^{-1}$), FBM = palatable monocot biomass ($\text{kg} \cdot \text{ha}^{-1}$), COVP = palatable species cover (%), DEGP = ratio of productive degradation-indicator species (%), DEGE = ratio of environmental degradation-indicator species (%), I-FOR = forage component index, GUA = guanaco density ($\text{OE} \cdot \text{ha}^{-1}$), LIV = livestock density ($\text{OE} \cdot \text{ha}^{-1}$), ANI = total animal density ($\text{OE} \cdot \text{ha}^{-1}$), PANI = potential animal density based on available forage ($\text{OE} \cdot \text{ha}^{-1}$), DMD = dry matter digestibility (%), NDF = neutral detergent fibre (%), MEC = metabolizable energy content (Mcal), CPC = crude protein content (%), I-ANI = animal component index, RIC = plant species richness (n), RICN = native species richness (n), RICE = exotic species richness (n), COV = total understory cover (%), COVD = dicot cover (%), COVM = monocot cover (%), COVE = exotic species cover (%), COVN = native species cover (%), EXO = ratio of exotic and total cover (%), I-BIO = biodiversity component index.

Finally, the biodiversity component index (I-BIO) showed significant differences among environments, where HF were lower than other environments (PF-DG-WG) (Table 2C). Total plant richness and native plant richness (RIC and RICN) were higher in forests (PF > HF) than in grasslands (DG > WG). However, richness of exotic species (RICE) was significantly lower in DG than other environments. Understory cover was higher in harvested stands (HF > PF > WG > DG), but when considering specific groups, dicot plants not showed significant differences (COVD), and monocotyledonous plants (COVM) were higher in WG than other environments. Cover of native species (COVN) was highest in WG compared to other environments, while cover of exotic species (COVE) was higher in forests (HF > PF) than in grasslands (WG > DG). Finally, the exotic species percentage (EXO) was higher in harvested stands (HF) than in other environments (PF > WG > DG).

Index comparison among environments

The cross between I-TREE and I-ENV showed a gradient from PF to DG, where HF occupied an intermediate position due to forest structure removal during harvesting (Fig. 2). WG significantly differed from other environments, due to the differences described earlier (Table 1). The cross between I-TREE and I-FOR showed that harvesting did not greatly influence over values, where HF was similar than PF, but split forested environments from openlands. Besides, harvested stands improved the animal index (I-ANI), reaching to similar values than WG. Finally, when comparing I-TREE and I-BIO, there was a significant reduction in biodiversity values in HF compared to other environments (WG > DG-PF) (Fig. 2).

Variations in soil profile among environments

Multiple ANOVAs allowed us to identify differences among the studied environments in SD, SM, and pH (Table 3), as well as significant differences soil organic carbon content (C) and the different organic matter components (POM and DOM). Carbon was higher in WG than in the other environments, where HF presented a slight decrease. The proportion of MOP and MOAM was consistent across the studied environments, except in DG, where MOP decreased and MOAM increased compared to other treatments. The presence of sand and rocks (ROC) did not follow a clear pattern, with PF > DG > HF > WG. Finally, the proportion of total roots, and roots of trees and shrubs in the soil (RTOT and RTRE) did not change among treatments, but roots of herbs and grass (RFOR) considerably increased in WG compared to other environments (HF > DG-PF).

Multiple ANOVAs also showed significant differences when soil depth gradient was analysed (Table 3). An increase in soil density (SD) and decrease in soil organic carbon (SOC) were observed with increasing soil depth. No significant differences were detected in SM, POM and DOM. Regarding soil acidity (pH), it was higher at the first layer (0–10 cm) compared to the second layer (10–20 cm) and increasing soil depth. The percentage of sand and rocks (ROC) also increased with soil depth, where total roots (R-TOT) was highest at the first soil layer (0–10 cm), and showed a second peak at 20–30 cm depth. This is due to a differential behaviour between forage plants (R-FOR) and woody plants (R-TRE), where the first ones significantly decreased beyond the second soil layer (> 20 cm depth). The roots of trees and shrubs presented a double distribution, occupying the first layer (0–10 cm) and then appearing more frequently at 20–30 cm. Besides, roots of both categories (R-FOR and R-TRE) were observed at the deeper layers.

Multiple ANOVAs also detected several interactions, which are presented in Fig. 3. Many of them are due to changes in the slope along the studied soil depth gradient, highlighting some soil peculiarities that differ among the studied treatments, e.g. an increase of SD with soil depth was observed, however, HF was higher than PF throughout the entire soil depth gradient. This could indicate that the impacts generated by harvesting resulted in changes much more significant than was expected. Besides, the soil nitrogen content (N) observed in dry grasslands (DG) only occurred at the first layer (0–10 cm), possibly due to the higher contribution of forage roots or animals faeces. Available phosphorus content (P) was higher at HF for almost the entire soil profile, except for the 20–30 cm layer in PF, where an increase in the quantity of tree and shrub roots (R-TRE) was observed.

Table 3

Multiple analyses of variance for soil characteristics along a depth gradient for the studied environments (PF = primary forests, HF = harvested forests, DG = dry grasslands, WG = wet grasslands).

Factor	Variable	SD	SM	pH	C	POM	DOM	ROC	RTRE	RFOR	RTOT
A: Environment	PF	1.04b	15.9a	4.6a	3.1a	80.6ab	19.4ab	14.2b	0.37	0.02a	0.39
	HF	1.23c	13.0a	5.1b	2.8a	83.7b	16.3a	6.4ab	0.43	0.10a	0.53
	DG	1.05bc	20.5a	5.3b	3.2a	77.1a	22.9b	12.6b	0.17	0.02a	0.19
	WG	0.40a	142.9b	4.9ab	18.8b	81.5ab	18.5ab	1.6a	0.20	0.71b	0.91
	F	57.78	64.15	13.83	86.56	3.95	3.95	5.75	0.58	14.28	2.43
	(p)	(< 0.001)	(< 0.001)	(< 0.001)	(< 0.001)	(0.013)	(0.013)	(0.001)	(0.627)	(< 0.001)	(0.073)
B: Soil depth (cm)	0–10	0.65a	51.5	5.0ab	11.9c	83.3	16.7	2.9a	0.63b	0.78b	1.41b
	10–20	0.93b	46.2	4.7a	7.8b	79.5	20.5	3.2a	0.11a	0.13a	0.24a
	20–30	0.97b	48.1	4.9ab	6.6b	79.9	20.1	5.9a	0.61b	0.08a	0.69ab
	30–40	1.02b	51.1	5.0ab	6.3b	78.5	21.5	11.3ab	0.10a	0.04a	0.14a
	40–50	1.08b	43.4	5.2b	2.2a	82.6	17.4	20.2b	0.01a	0.03a	0.04a
	F	9.81	0.15	2.90	19.84	1.44	1.44	7.15	2.76	10.64	6.70
	(p)	(< 0.001)	(0.964)	(0.030)	(< 0.001)	(0.235)	(0.235)	(< 0.001)	(0.035)	(< 0.001)	(< 0.001)
A x B	F	0.98	0.17	3.69	2.71	2.45	2.45	2.10	0.82	7.23	2.35
	(p)	(0.476)	(0.999)	(< 0.001)	(0.006)	(0.012)	(0.013)	(0.030)	(0.634)	(< 0.001)	(0.015)

F = Fisher's test, p = probability where different letters indicate differences by Tukey test (p < 0.05). SD = soil density (g.cm³), SM = soil moisture (%), pH = soil acidity, C = soil organic carbon (%), POM = particulate organic matter (%C), DOM = dissolved organic matter (%C), ROC = rocks and sand (% soil), RTRE = tree and shrub roots (% soil), RFOR = herb and grass roots (% soil), RTOT = total roots (% soil).

The percentage of soil organic carbon (C) followed an expected pattern, decreasing along the profile, much more abruptly in forests (PF-HF) and DG than in WG, where the content remained high even at greater depths (< 40 cm). The ratio of POM and DOM showed different patterns among the studied environments, e.g. HF maintained a consistent ratio across the profile, while PF and DG decreased POM and increased DOM with soil depth, and then decreased again. In WG, POM increased and DOM decreased at the second layer compared to the first and last layers. Sand and rock percentage (ROC) increased with soil depth in most of the treatments (PF, HF, DG), while in WG they were only observed near the surface (0–10 cm). Finally, the roots (R-TRE and R-FOR) followed the previously described pattern, and clearly showed a differential behaviour between the forests and grasslands. Besides, the peak of R-TRE at 20–30 cm was higher in HF than PF, even though the number of trees in the harvested areas was significantly lower, indicating a greater potential positive synergy between components (e.g. trees and forage).

Discussion

The SSP in fire forests is based on continuous forest interventions (e.g. thinning), which enhances the forage component and also increases the growth rates of remnant trees, promoting diverse forest uses including timber production (Martínez Pastur et al. 2018). These cuttings must be implemented maintaining the forest resilience (Peri et al. 2022), due to if they exceed the ecological thresholds (Peri et al. 2017b) can lead to new permanent hybrid or de-novo ecosystems (Frangi et al. 2015). It has been proposed that the magnitude of these cuttings must consider local hydrological regimes (e.g. Peri et al. 2016, 2017a) and their ecosystem and biodiversity values (e.g. Martínez Pastur et al. 2017, 2020). However, these SSP proposals are not inflexible and must be adapted to the production strategies, aiming to diversify the different commercial products to increase the resilience of the ranches (Peri et al. 2021, 2022).

Silvicultural treatments in the framework of SSP require natural regeneration to ensure the persistence of the tree components over time (Soler et al. 2013; Bahamonde et al. 2018). It is expected that the increased availability of limitant resources in harvested stands (e.g. light and rainfall) could favour natural regeneration (e.g. seeds or sprouts). However, our results showed lower values than in unmanaged forests. This can be due to negative synergies (e.g. browsing and exotic plant competition) (Soler et al. 2012), and must be monitored to determine the need of active actions (e.g. assisted planting or temporary protections) (Peri et al. 2016). However, SSP also are related to other losses, e.g. our study showed a significant increases of soil density, which can increase nutrient contents (weight/volume) but not necessarily the effective percentages (e.g. loss in soil C percentage) (Martínez Pastur et al. 2021, 2022). However, these impacts can produce positive synergies through the introduction of plants that fix nitrogen or through an increase in animal faeces in the forest soil (Sitters et al. 2020; Kacprzak et al. 2023).

Our results showed a beneficial effect of trees and shrubs through the functions of root distribution across the soil profile, where an increase in the deeper layers was observed as previously reported (Peri et al. 2008; Reyna-Bowen et al. 2020). Contrary, herbs and grasses mostly occupied the surface layers, although in both cases, some roots reached to deeper layers. SSP offered the best balance including the higher proportion of roots from the forage and tree components compared to unmanaged forests. Deep roots allowed greater uptake of water and nutrients from lower layers, and through the leaf decomposition, benefit to the forage component, as previously cited (Knowles 1991; Sales-Baptista and Ferraz-de-Oliveira 2021).

Despite the greater forage supply of openlands (DG and WG), there were no differences in the animal component. This underscores the importance of fire forests as suitable environments for livestock uses, both on managed and unmanaged stands within productive landscapes. The protective and comfort role provided by trees to animals (Lemes et al. 2021) is generally underestimated and rarely evaluated in SSP. It is necessary to promote these kind of values, and include them to improve livestock management proposals and design of pasture strategies in the ranches. Both unmanaged and harvested forests maintained high palatability of the forage component, which was comparable to wet grasslands (WG), mainly due to the maintenance of the assembly and diversity of plant species. However, dry grasslands (DG) offer lower-quality forage, but it is available during the winter season (Posse et al. 1996). Finally, it is noteworthy that biodiversity values were lower in the harvested forests (HF) compared to other environments, which mostly maintained their ecological integrity. This is the main disadvantage of SSP at Southern Patagonia, which must be assessed and redesigned to maintain acceptable levels of biodiversity, reducing loss ecosystem functionality and identity (Peri et al. 2022). Moreover, the introduction of more palatable exotic species should be carried out with precautionary principles (HF had a higher proportion of exotic species cover), especially because it can disrupt the natural resilience of these ecosystems (Quinteros et al. 2010), favouring the invasion of unwanted species (e.g. *Hieracium pilosella*) (Martínez Pastur et al. 2020; Cooke et al. 2021).

The studied environments differed in the provision of ecosystem services, both in type and magnitude, and varied in their values associated to economic and conservation, e.g. it was observed a differential resistance to species invasions and degradation processes (e.g. livestock impacts). Besides, it was observed differential values of nutrient recycling and soil characteristics to face factors of change (e.g. different ratios and contents of POM and DOM across the soil profile), which make them more resistant, e.g. erosion vulnerability (Kumar et al. 2023). SSP promotes changes in forests under natural dynamics, creating a balance of losses (e.g. tree and biodiversity components) and gains (e.g. environmental and forage components) to offer to the animal component better resource availability, increasing productivity in terms of quantity and quality, as well as promoting the diversification of commercial products (e.g. high-quality timber). In this framework, SSP promotes the establishment of new productive environments with intermediary characteristics between natural forests and grasslands, as was showed in the obtained results.

Conclusions

SSP in Tierra del Fuego improves some of the productive indicators related to quality and quantity of the forage component, generating functional changes in soils that generate multiply ecosystem benefits. However, these benefits result from the loss of other ecosystem and biodiversity values. These positive and negative synergies can be regulated through different management strategies to reduce the observed impacts and to maintain ecosystem resilience. Additionally, it is necessary to better determine which changes observed throughout the soil profile (0–50 cm) are reversible or persistent in the long-term. SSP contributes to increasing the forest productivity in monetary (e.g. livestock stock) and non-monetary or indirect values (e.g. animal comfort), combining multiple objectives in a single paddock. The results presented here contribute to identifying potential indicators of each component involved in SSP. The measurement of these indicators can contribute for effective long-term monitoring and the development of better adaptive management strategies.

Declarations

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Figures

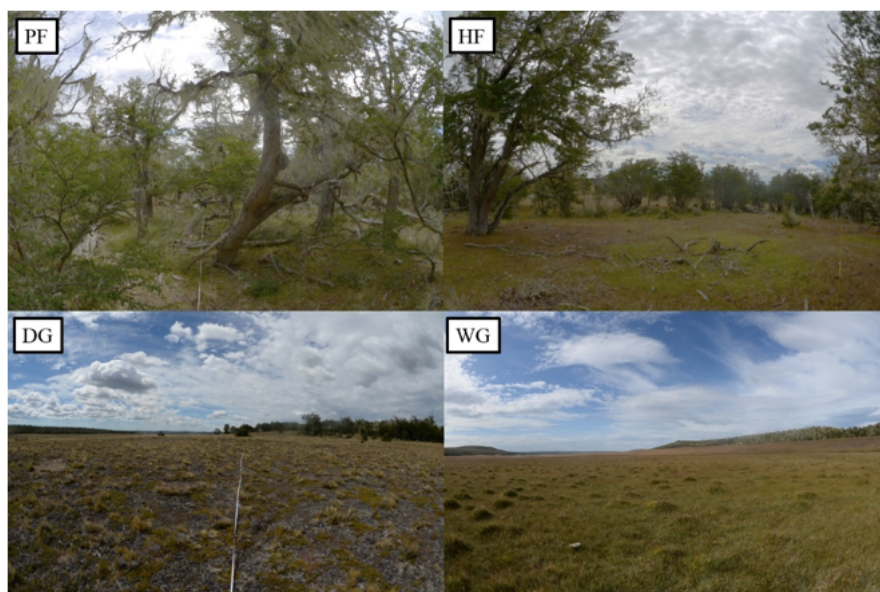


Figure 1

Studied environments at El Roble ranch. PF = primary forests, HF = harvested forests, DG = dry grasslands, WG = wet grasslands.

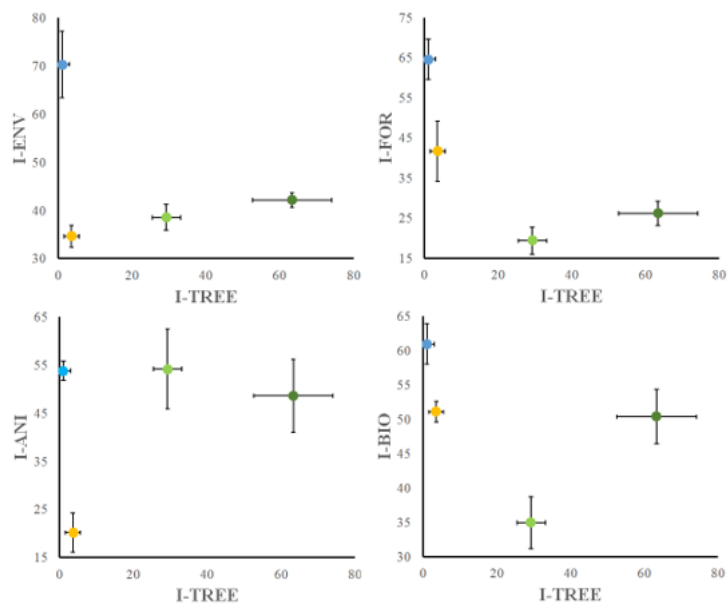


Figure 2

Comparative indices (mean and standard error) of the different components analysed for the studied environments (dark green = primary forests, light green = harvesting forests, orange = dry grasslands, light blue = wet grasslands). I-TREE = tree component index, I-ENV = environmental component index, I-FOR = forage component index, I-ANI = animal component index, I-BIO = biodiversity component index.

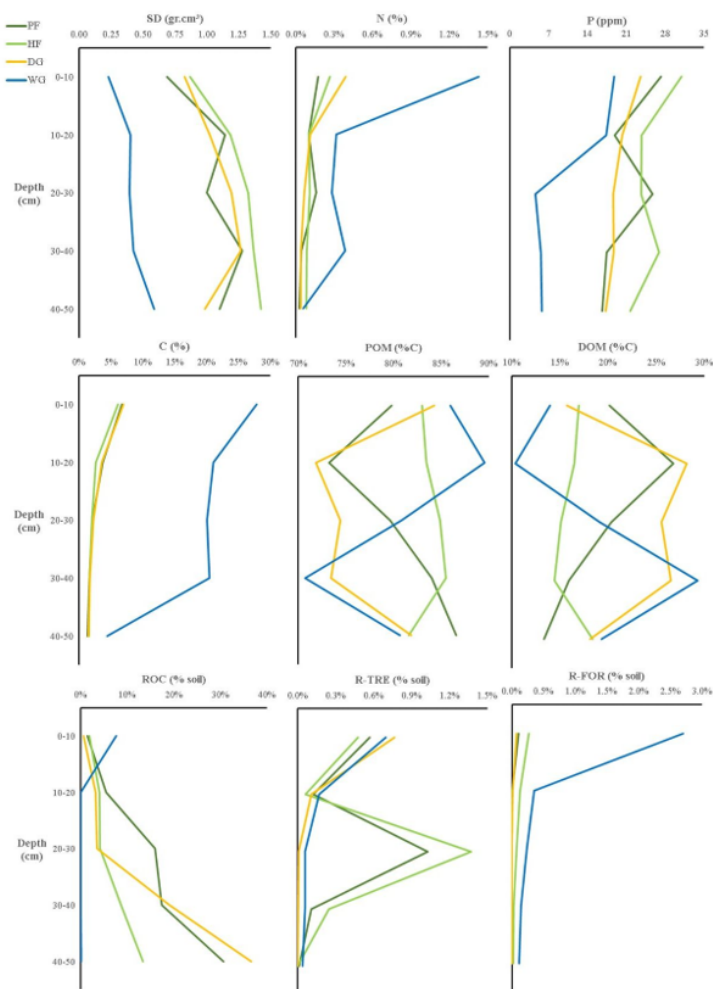


Figure 3

Relationships among soil characteristics and soil depth for the studied environments (PF = primary forests, HF = harvested forests, DG = dry grasslands, WG = wet grasslands). SD = soil density (g.cm^3), N = soil nitrogen content (%), P = soil phosphorus content (ppm), C = soil organic carbon (%), POM = particulate organic matter (%C), DOM = dissolved organic matter (%C), ROC = rock and sand content (% soil), RTRE = tree and shrub roots (% soil), RFOR = herb and grass roots (% soil).