

Carbon sequestration in semi-arid rangelands: Comparison of *Pinus ponderosa* plantations and grazing exclusion in NW Patagonia

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Abstract

The large global extension of arid and semi-arid regions together with their widespread degradation give these areas a high potential to sequester carbon. We explored the possibilities of semi-arid ecosystems to sequester carbon by means of rangeland exclusion and afforestation with *Pinus ponderosa* in NW Patagonia (Argentina). We sampled all pools where organic carbon accumulates in a network of five trios of adjacent grazed, non-grazed and afforested stands (age: 12–25 years, density 605–1052 trees ha⁻¹). After 15 years since trees were planted, afforestation added ~50% more C to the initial ecosystem carbon pool, with annual sequestration rate ranging 0.5–3.3 Mg C ha⁻¹ year⁻¹. Carbon gains in afforested stands were higher above than below-ground (150% vs. 32%). Root biomass differences (374% more in afforested vs. grazed stands, $p = 0.0011$) explained below-ground carbon contrasts whereas soil organic carbon showed no differences with afforestation. By contrast, grazing exclusions did not result in significant changes in the total carbon storage in comparison with the adjacent grazed stands ($p = 0.42$) suggesting a slow ecosystem recovery in the time frame of this study (~15 years of exclusion). Nevertheless, higher litter amount was found in the former (+53%, $p = 0.07$). Neither, soil organic carbon nor root carbon showed significant differences between grazed and non-grazed conditions. Considering that more than 1.1 millions of hectares of the studied ecosystems are highly degraded and suitable for tree planting, afforesting this area could result in a carbon sequestration rate of 1.7 Tg C year⁻¹, almost 6% of the

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current fossil fuel emissions of Argentina; however environmental consequences which could emerge from this deep land use shift must be taken into account when afforestation program are being designed.

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1. Introduction

Low and highly variable precipitation inputs constrain plant productivity in arid and semi-arid regions, limiting economic alternatives to grazing activities where irrigation is not feasible (Noy-Meier, 1973). The traditional management of these rangelands, often associated with stocking density over the carrying capacity, has resulted in floristic and physiognomic changes, losses of soil organic carbon, increase of bare soil and eventually desertification (Golluscio et al., 1998; Lal, 2002). Restoration attempts have been hardly implemented because the low economic return per unit area of these systems makes them a risky investment (Glenn et al., 1998). The emergence of a prospective carbon market, as a tool to promote carbon sequestration and offset the increasing atmospheric carbon dioxide concentration, may offer new incentives to restore arid ecosystems, especially in those places where land degradation has led to a decline in economic yields (Prince et al., 1998).

In recent years arid and semi-arid regions, hereafter drylands, have been regarded as potential carbon sinks (e.g. Squires, 1998; Lal, 2002; Ardö and Olsson, 2003; Grünzweig et al., 2003). Drylands cover ~45% of the global land surface and, despite their low soil organic carbon (SOC) concentration; encompass ~16% of the global soil C pool (Ojima et al., 1993; Jobbágy and Jackson, 2000). Desertification, affecting more than two-thirds of drylands, has likely caused carbon losses of ~20–30 Pg globally (Ojima et al., 1995; Lal et al., 1999). Restoring these systems through the adoption of appropriate land use practices could yield significant ecosystem carbon gains. Large global extension and widespread degradation give drylands one of the highest potential to sequester carbon. A potential global sequestration of ~1 Pg C year⁻¹, or ~30% of current increases in the atmospheric CO₂ concentration (Squires, 1998; IPCC, 2001), denotes the relevance of drylands in the fate and management of the global carbon cycle.

Besides the possibility of atmospheric CO₂ removal and its potential commercialization, higher carbon pools in soils and vegetation entail associated benefits that enhance the overall productivity of drylands. SOC gains improve soil quality through better water-holding capacity, fertility and biodiversity (Haynes and Naidu, 1998; Loveland and Webb, 2003; Evrendilek et al., 2004) and have a stabilizing effect on soil structure that can prevent erosion (Boix-Fayos et al., 2001). Carbon gains in plant and litter pools are often associated to runoff reduction and infiltration and soil moisture increase (Rostagno et al., 1991; Abril and Bucher, 2001). In addition, lower soil temperatures, associated with higher ground cover, act decreasing the decomposition rate of the soil organic matter (Archer, 1995).

Two of the most widely suggested options to sequester carbon in drylands are afforestation (tree planting) and rangeland restoration through grazing exclusion. Ecosystems sequester carbon when their uptake increase and/or their losses decrease. Afforestations can increase C influx through a higher and more efficient use of resources

for primary production. For example, the deep root distribution of trees (Schenk and Jackson, 2002) can allow the absorption of soil water at depths that are not accessed by the existing rangeland plants species. Grazing exclusion is likely to increase C uptake where overgrazing has reduced plant cover and/or impaired soil fertility (Oesterheld et al., 1999). In addition to C uptake shifts, afforestation and grazing exclusion can favour C sequestration through the reduction of C losses if higher ground cover reduces SOC decomposition and soil erosion. The slower C turnover rates associated with wood carbon allocation and the cessation of biomass removal by livestock may also decrease C losses in afforestations and exclosures, respectively (McIntosh et al., 1997; Schlesinger, 1997).

Some uncertainties make the outcome of afforestation and grazing exclusion on the C balance of dryland ecosystems hard to predict. The effects of afforestation on carbon pools have been largely examined in humid regions (Martin et al., 2001) but afforestation effects in drylands remain poorly explored. Woody encroachment of drylands suggests a strong effect of precipitation on SOC shifts (Jackson et al., 2002). Additionally, while high C amount can be store in wood biomass, this represents a labile pool, prone to leakage when harvested or burned. In rangelands, above-ground biomass and litter have been proved to increase after grazing removal (McIntosh et al., 1997), but SOC, the largest C pool in these systems, has shown non-conclusive response to grazing, with studies showing increases (Schuman et al., 1999; Weinhold et al., 2001; Reeder and Schuman, 2002), decreases (Bauer et al., 1987; Dormaar et al., 1989) and lack of changes (Milchunas and Lauenroth, 1993; McIntosh et al., 1997). The dynamics of shrub cover after livestock exclusion adds uncertainties to this land use modification.

Patagonia, in the southernmost portion of South America, is a cold temperate and semi-arid area ($\sim 700,000 \text{ km}^2$) dominated by semideserts and grass/shrub steppes (Leon et al., 1998). Since the introduction of domestic herbivores (mainly sheep) in the early 1900s, grazing has become the dominant land use. Overgrazing had triggered a widespread desertification process (Soriano and Movia, 1986; Paruelo and Aguiar, 2003). Grass steppes, with a total plant cover of $\sim 70\%$, were transformed into semideserts dominated by cushion-like shrubs with a total plant cover lower than 25% (Paruelo et al., 2004b). Changes in vegetation led to profound changes in soil due to water and wind erosion. In the last decades tree planting has been strongly supported by the Argentine government, resulting in $\sim 70,000 \text{ ha}$ of ponderosa pine plantations in the region with a current expansion rate of $\sim 5000 \text{ ha}$ per year (Laclau, 2003). Although grazing exclusion is a very uncommon practice in the region, the protective fencing of afforested paddocks has enclosed rangeland sections interrupting the access of domestic herbivores at many locations. This special array of adjacent land use situations (grazed and exclosed rangeland and tree plantations) provides a useful setting to explore the effects of afforestation and grazing exclusion on ecosystem carbon pools.

Here we explore the potential of semi-arid ecosystems to sequester carbon by means of afforestation with *Pinus ponderosa* (Dougl.) Laws and rangeland exclusion guided by the following hypotheses: (1) Afforestation increase ecosystem carbon pools through large biomass C gains. (2) Grazing exclusion increase ecosystem carbon pools through the combination of slight biomass, litter, and SOC gains. We tested these hypotheses in the north-west region of the Patagonian steppe using a network of five sites with adjacent grazed, non-grazed, and afforested stands in which above- and below-ground C pools were measured. The simultaneous and contiguous availability of these three situations allowed

us to apply a space-for-time substitution approach, whereby the grazed steppe was considered as the original situation before the land use modification.

2. Materials and methods

2.1. Study sites

The study was conducted in the vicinity of Maitén (42°06'S, 71°10'W; Chubut Province, Argentina) in the north-western Patagonia. Mean annual temperature and precipitation are 6 °C and 424 mm, respectively (Paruelo et al., 2000). Precipitation inputs are mainly concentrated in winter and fall (73%) and most events have < 5 mm (Jobbágy et al., 1995). The ratio of mean annual precipitation and potential evapo-transpiration is 0.40 (Paruelo et al., 1998) which corresponds to the semi-arid category according to Le Houérou (1996). Westerly winds represent between 65% and 75% of daily observations and the mean annual value of speed varies from 15 to 22 km h⁻¹ (Paruelo et al., 1998). Soils in the study area are derived from glacial and volcanic materials and have an important proportion of gravel and pebble stones throughout their profiles (Soriano, 1983). Topography is gentle with elevation ranging from 650 to 780 m. Study sites correspond to the transition between the Subandean and the Occidental floristic districts (Soriano, 1956; Leon et al., 1998), a grass-shrub steppe in which long-term grazing has led to plant cover reduction accompanied by primary productivity and carrying capacity decline (Aguiar et al., 1996; Paruelo et al., 2004a). In these particular sites the shrubs *Mulinum spinosum*, *Adesmia campestris*, *Berberis heterophylla*, *Colletia histrix* and *Senecio filaginoides* became dominant. Most of the grass biomass is accounted for by species of *Stipa* (*Stipa speciosa* and *Stipa humilis*). More mesophitic and palatable grasses (*Festuca pallescens*, *Poa ligularis*, and *Bromus pictus*) are important (or even dominant) in the non-degraded states of the steppes but they show a sharp decline under overgrazing conditions. Forbs usually account for less than 1% of total cover.

In an attempt to characterize the changes in carbon pools after afforestation and grazing enclosure we selected five sites located on a homogeneous physiographic unit (the El Maitén plain) (Sites: A, B, C, D and E). Each site included three land uses: grazed, non-grazed and ponderosa pine plantation. All stands in each trio were located on the same topographic position and shared the same soil type, as confirmed by direct observation in the field. Soils are coarse textured and highly drained. Gravels are found on the surface and stones below. Salinity/sodicity problems are not observed. Slopes ranged from 0.5% to 2.5% and soil cover was generally below 50%. Grazing exclusions were established at the same time trees were planted. In sites D and E suitable non-grazed stands were absent and only grazed and afforested stands were evaluated. The areas show clear signs of long-term degradation by overgrazing (R.A. Golluscio, pers. comm.).

Tree plantations ranged from 8 to 16 ha in size and were established between 1979 and 1992 (12–25 years of age at the time of our measurements) with planting densities of 1111 and 1600 trees per hectare (Table 1). No site disturbance preceded tree planting, except for hole digging in each tree position. Plantations were never harvested, fertilized nor irrigated. Plantations were thinned and pruned between 10 and 11 years of age, and the residues were chipped and dispersed in the ground.

Table 1
Characteristics of *Pinus ponderosa* plantations

Site	Age (year)	Plantation size (ha)	Density (trees ha ⁻¹)	DBH (cm)	BD (cm)	H (m)
A	12	16	679.0	7.8	13.8	3.6
B	12	16	703.7	9.3	15.6	4.0
C	14	8	604.9	9.2	15.3	4.1
D	15	10	716.0	16.9	24.7	6.8
E	25	10	1051.7	17.3	23.7	6.9
Average	15.6	12	751.1	12.1	18.6	5.1

The age, size, density, diameter at breast height (DBH), basal diameter (BD) and height (*H*) of afforested stands are indicated.

2.2. Data collection and analysis

In order to characterize carbon storage and distribution we sampled all pools where organic carbon accumulates. To evaluate carbon stocked in tree compartments we developed forest biomass equations. Power functions ($y = ax^b$) relating dry weight of each compartment with diameter at breast height (DBH) and basal diameter (BD) were established. Regression equations were log-linear transformed and the significance of the *b* coefficient ($H_0: b = 0$) was tested. Ten trees ranging 0–23 cm of DBH distributed across several plantations were cut down and foliage, cones, branches, stem and roots were weighed separately. Subsamples of each compartment were oven-dried at 65 °C to determine water content. In the case of roots we worked with five trees establishing a 90° section of circle of 1.20 m diameter around the stem in which all roots found to a depth of 2 m and retained in a 1 cm sieve were collected. Since higher correlations coefficients were obtained using BD, we used this variable to estimate tree biomass at stand level (Table 2). Forest inventory was performed in rectangular plots of 270 m², with the exception of site E where, due to irregular tree distribution, the nearest individual method was used (Barbour et al., 1987). Three replicates per site were established and basal and breast height diameters were recorded for every tree.

Litter, as well as grass, shrubs and forbs biomass was collected in rectangular 2 m² plots (0.2 × 10 m²). Subsamples of each compartment were oven dried and the dry matter coefficients were used to transform fresh weight into dry matter. To account for those roots located outside the circle in tree plantations and the ones located in non-afforested stands, soil samples with known volume were collected down to a depth of 2 m (0–5, 5–10, 10–30, 30–50, 50–100, 100–150, 150–200 cm) and sieved (1 cm). Gravel and roots were separated. Gravel was weighed in the field and roots were taken to the laboratory and weighed after oven-drying. A standard coefficient of 0.5 (Brown and Lugo, 1984; IPCC, 1996) was used to obtain the biomass carbon content. Carbon stored in soil organic matter and bulk density were quantified to a depth of 2 m with the same depth intervals used for root sampling. Four samples were taken from each depth and pooled for lab analysis. All soil samples were air-dried and sieved (2 mm) prior to determination of total organic C using the Walkley–Black method. Statistical analyses were conducted using INFOSTAT 1.1 software (Universidad Nacional de Córdoba, Argentina, 2002). The statical model used was incomplete blocks (Hinkelmann and Kempthorne, 2005) and differences between tree plantations, grazed and non-grazed steppe were analysed using two-way ANOVA considering each site as a block.

Table 2
Forest biomass equations used to calculate carbon storage in tree stands

Compartment	Number of trees	Predictive variable							
		Diameter at breast height (cm)				Basal diameter (cm)			
		Coef <i>a</i>	exp <i>b</i>	<i>r</i> ²	<i>p</i>	Coef <i>a</i>	exp <i>b</i>	<i>r</i> ²	<i>p</i>
Stem	10	201.41	1.71	0.95	<0.0001	4.22	2.72	0.98	<0.0001
Branches	10	130.61	1.63	0.70	0.0095	2.45	2.71	0.90	<0.0001
Foliage	10	197.71	1.39	0.73	0.0069	9.91	2.17	0.91	<0.0001
Coarse roots	5	272.81	1.01	0.84	0.0840	29.26	1.54	0.95	0.0054
Taproot	5	170.62	1.39	0.87	0.0670	3.05	2.43	0.98	0.0013

Biomass of each tree compartment (stem, branches, foliage, coarse roots and taproot) was related with diameter at breast height (DBH) and basal diameter (BD) through power functions ($y = ax^b$). The resulting dry weigh is expressed in grams. The determination coefficient (r^2) as well as the *p*-value of the *b* coefficient after log-transformation is indicated. BD and DBH were related by $BD = 1.2DBH + 4.6$ ($r^2 = 0.95$, $p < 0.0001$).

Table 3
Carbon distribution ($Mg\ ha^{-1}$) among pools in grazed, non-grazed and afforested stands

	Grazed		Non-grazed		Afforested	
	Mean	S.E.	Mean	S.E.	Mean	S.E.
Grasses	1.52	0.72	0.53	0.05	0.22	0.11
Forbs	0.25	0.13	0.17	0.04	0.11	0.05
Shrubs	5.07	2.49	2.47	0.54	1.57	0.69
Trees	0.00	0.00	0.00	0.00	12.29	4.57
Litter	1.00	0.14	1.69	0.13	5.39	2.65
Total above-ground carbon	7.84	2.06	4.86	0.64	19.58	7.79
Roots	2.60	1.19	1.58	0.64	12.33	1.79
Soil organic carbon	33.88	3.44	35.36	5.42	35.79	5.85
Total below-ground carbon	36.48	3.96	36.94	5.32	48.12	8.60
Total carbon	44.33	5.58	41.80	4.76	67.71	13.95

The mean value and standard error are indicated ($n = 5$ for grazed and afforested stands; $n = 3$ for non-grazed stands).

3. Results

Afforested stands presented significantly higher carbon storage compared to the adjacent grazed steppe ($p = 0.04$, Table 3). After 15 years since trees were planted, afforestation added $\sim 50\%$ more C to the initial ecosystem carbon pool. Annual C sequestration rates averaged $1.5\ Mg\ ha^{-1}\ year^{-1}$ (range $0.5\text{--}3.3\ Mg\ C\ ha^{-1}\ year^{-1}$) with C stock gains averaging $23\ Mg\ C\ ha^{-1}$ (range $6\text{--}50\ Mg\ C\ ha^{-1}$) (Fig. 1).

Carbon gains in afforested stands were higher above than below-ground (150% vs. 32%). With marginally significant contrasts, grass and shrub carbon pools declined (-85% , $p = 0.10$; -70% , $p = 0.09$, respectively) and litter carbon increased in afforested

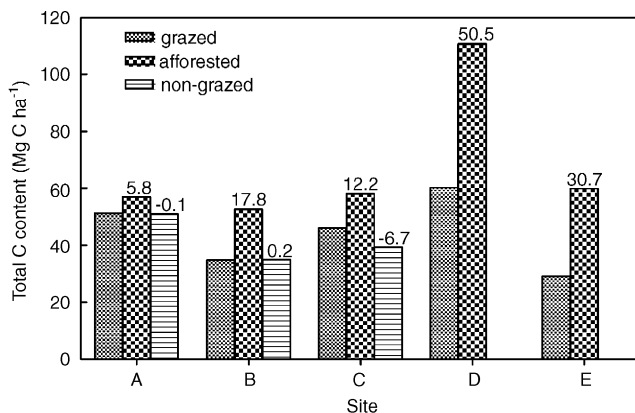


Fig. 1. Total carbon stored in each sampled stand. Numbers above afforested and non-grazed treatments indicate their differences with grazed stands (Mg C ha^{-1}).

stands (+439%, $p = 0.06$). Forbs were a minor component of total carbon pools and showed higher (+127%) absolute abundance in grazed steppe stands. Root biomass differences (374% more in afforested vs. grazed steppe stands, $p = 0.001$) explained below-ground carbon contrasts whereas SOC showed no differences with afforestation.

Grazing exclusions did not differ in their total carbon storage from adjacent grazed stands ($p = 0.42$, Table 3). Nevertheless, higher litter amount was found in the former (+53%, $p = 0.07$). Neither, SOC nor root carbon showed significant differences between grazed and non-grazed conditions. Differences in total C storage attribute to grazing exclusion varied between -6.7 and 0.2 Mg C ha^{-1} (Fig. 1).

Grazed and non-grazed stands had most of their carbon below-ground, with SOC representing 76% and 84% of the total ecosystem carbon, respectively (Fig. 2). In afforested stands the highest carbon storage was also allocated below-ground, but SOC contribution decreased to 53%. Litter and biomass carbon contribution was less than 4% and 20% in non-afforested stands, respectively, and were doubled after afforestation (Fig. 2).

The relative contribution of shrub and grass to the total above-ground biomass carbon did not differ between grazed and non-grazed stands (76–20% for shrub and grass C, respectively), whereas forbs showed higher contribution in the latter (5% vs. 1%, $p = 0.006$). In afforested stands, the contribution of grass and herbs was negligible and the highest percentage of above-ground biomass carbon was associated with trees, accounting for 87%.

The vertical distribution of SOC did not differ across treatments, with the first meter contributing $\sim 94\%$ of the total SOC measured down to two meters. Roots, in contrast, were more deeply distributed in afforested stands, compared with grazed and non-grazed steppe stands (Fig. 3). The relative root contribution of the top 10 cm of the soil profile to the total pool of roots measured was: grazed stands, 24% (0.6 Mg C ha^{-1}); non-grazed, 26% ($0.41 \text{ Mg C ha}^{-1}$); and afforested stands, 4% (0.5 Mg C ha^{-1}). These patterns reversed at depth with root contribution from the 1.5–2 m depth interval being: grazed stands, 0.6% ($0.01 \text{ Mg C ha}^{-1}$); non-grazed stands, 4% ($0.06 \text{ Mg C ha}^{-1}$); and afforested stands, 9%

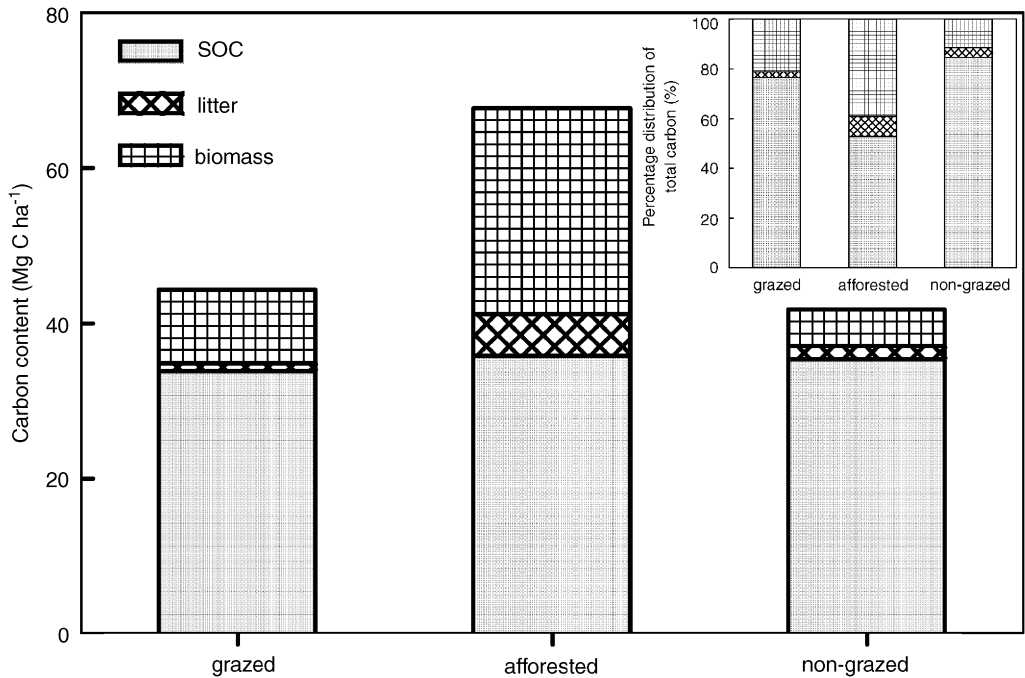


Fig. 2. Carbon distribution (Mg C ha^{-1}) among soil organic carbon (SOC), litter and biomass in grazed, non-grazed and afforested stands. Bars correspond to the average value of five sites (grazed and afforested stands) and three sites (non-grazed stands). The inset graphic represents the percentage carbon distribution among the same pools.

(1.1 Mg C ha^{-1}). Notably, root carbon observed in the second meter of afforested stands (3.6 Mg C ha^{-1}) exceed greatly all root carbon observed throughout the soil profile (0–2 m) in non-afforested ones ($2.09 \text{ Mg C ha}^{-1}$).

4. Discussion

While the conversion of grass/shrub steppes to ponderosa pine plantations in Patagonia resulted in significant ecosystem carbon gains, grazing exclusion yielded no changes (Table 3). The average carbon sequestration rate of plantations that we observed ($1.5 \text{ Mg C ha}^{-1} \text{ year}^{-1}$) matched observations from other afforested systems in semi-arid regions (Grünzweig et al., 2003) and evidenced the potential of afforestation for carbon sequestration programs in strongly water-limited ecosystems. Changes in carbon pools after tree planting were mainly due to tree biomass and litter increases, with SOC pools remaining unaffected. Higher litter amount and forb contribution were the only detectable changes after grazing exclusion, and they did not translate into higher total carbon storage.

A large fraction of the Patagonian grass and grass–shrub steppes (Subandean district, areas receiving more than 300 mm of mean annual precipitation) has experienced a profound degradation due to overgrazing (Paruelo and Aguiar, 2003; Paruelo, 2005). Evidences from areas where herbivores were excluded for >10 year indicated that

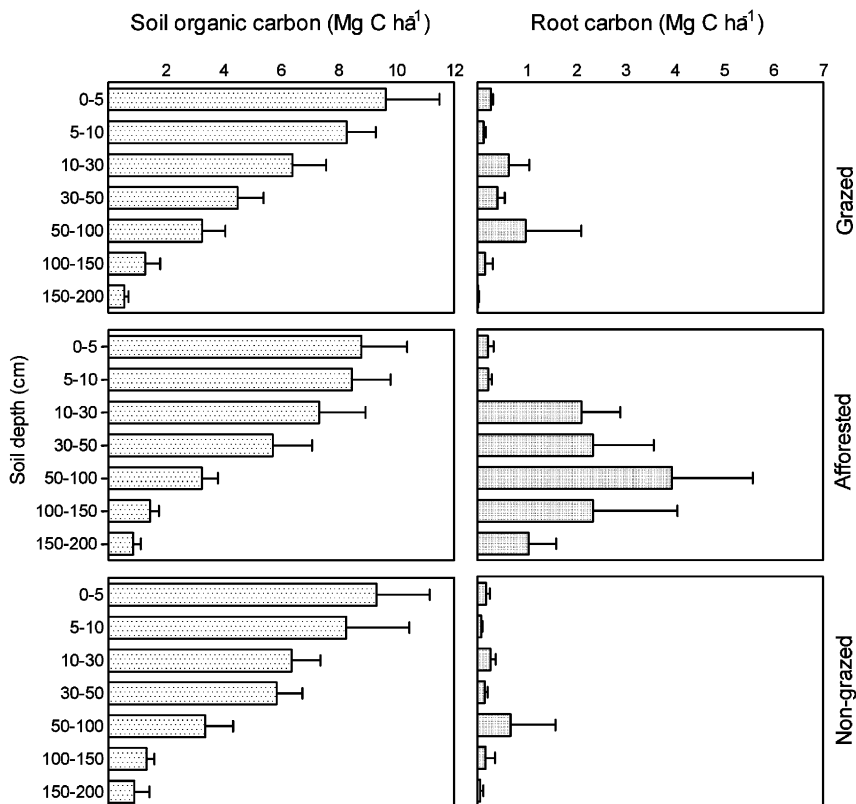


Fig. 3. Distribution of soil organic carbon (SOC) and root carbon (Mg C ha^{-1}) in grazed, non-grazed and afforested stands (mean + S.E.) with depth. Bars indicate the average value of five sites (grazed and afforested stands) and three sites (non-grazed stands).

restoration of the original steppe is very unlikely, at least in the short-term (Paruelo and Aguiar, 2003; Paruelo, 2005). Forage productivity and conservation value of the degraded steppes is low and afforestation appears as an alternative to increase the capacity of this area to provide ecosystems and economic services and goods. Estimates for a representative portion of the district suggest that a 22% is highly degraded (Paruelo, 2005). Extrapolating this figure to the whole Subandean district yield an area of 1.12 million ha. Afforesting this area could result in a carbon sequestration rate of $1.7 \text{ Tg C year}^{-1}$, almost 6% of the current fossil fuel emissions of Argentina (Marland et al., 2003).

Although promising, these sequestration estimates should be treated with caution because some factors add uncertainties to the net outcome. The sequestration rate that we estimated was based on young pines plantations (~ 15 years of age), characterized by high initial growth rates (Cochran and Barret, 1998). Considering that plantations are clearcut 35–40 years after planting (Laclau et al., 1999), growth and C sequestration rate decay could be expected as plantations get older (Binkley et al., 2002). Our estimate does not take into account costs associated with the planting and harvesting stages, potential carbon losses from disturbance (storms, pests and fires) or post-harvest carbon losses in timber use, which certainly modify sequestration estimates (Jackson and Schlesinger, 2004).

Another controversial factor is the final fate and C turnover of forest products (e.g. building materials vs. firewood). However, even when biomass is used as energy source, it would be offsetting atmospheric CO₂ increases by replacing fossil fuels, protecting existing natural forests resources at the same time (Albrecht and Kandji, 2003).

An enhancement of primary productivity together with decrease biomass turnover is probably explaining the higher carbon storage observed in afforested stands. Considering a very conservative estimate of above-ground net primary productivity (ANPP) of afforested stands (above-ground trees biomass + litter/plantation age) yields an average productivity rate of 215 g dry matter (DM) m⁻² year⁻¹ which falls within the 100–600 g m⁻² year⁻¹ production range considered for semi-arid zones (Noy-Meier, 1973). Published models relating ANPP in grassland/steppe ecosystems to annual precipitation (Lauenroth, 1979; Sala et al., 1988) predict an average value of 209 g DM m⁻² year⁻¹ for the study area, and measurements from similar ecosystems range between 80 and 270 g DM m⁻² year⁻¹ (Soriano, 1983; Defossé and Bertiller, 1991; Fernández et al., 1991). These values correspond to relatively undisturbed sites. Satellite-based estimates of ANPP for the study area yield values ranging between 60 and 112 g DM m⁻² year⁻¹ (Paruelo et al., 2004b). These rough comparisons suggest that afforested stands had at least doubled the ANPP of their hosting systems approaching or exceeding the levels of non-degraded steppes.

A more efficient use of rainfall by trees could be leading to higher productivity rates in afforested stands. Soil water simulation modelling has shown that drainage, defined as water reaching a depth of 200 cm, accounts for up to 36% of total water inputs in the Patagonian steppe (Paruelo et al., 2000). Afforested stands, with more roots reaching this depth than non-afforested ones (Fig. 3), would probably present more opportunities to convert rainfall into productivity. However, it is interesting to highlight that this drainage water is the main source for meadows located in small valleys (Paruelo et al., 2000), which make, despite their low areal coverage, an important contribution to the landscape primary productivity (Paruelo and Golluscio, 1994; Paruelo et al., 2004b) and a critical resource for the livestock (Texeira and Paruelo, 2006). Therefore, while tree planting could increase the productivity of the afforested stand, the productivity of the region as a whole may remain unaffected or even decrease.

In contrast with our initial expectations, grazing exclusion did not increase ecosystem carbon pools (Table 3). However, these results are not completely surprising and have been observed in other regions (Milchunas and Lauenroth, 1993; McIntosh et al., 1997). Two possible explanations for the lack of carbon gains following grazing exclusion are low grazing intensity in the initial condition and/or slow ecosystem recovery in the time frame of this study (~15 years of exclusion). The high proportion of shrubs observed in grazed stands (>70% of the total above-ground biomass, Table 3) would indicate an intensive grazing situation in a region that is dominated by grasses in its least disturbed stage (Leon and Aguiar, 1985), making slow recovery the most likely cause of poor carbon gains in the exclosures that we studied (Soriano et al., 1980; Valone et al., 2002). Long lags in rangeland recovery could be associated with a limited availability of plant propagules (Soriano and Sala, 1986). Indirect effects of grazing that remain many years after livestock removal, e.g. soils may be prone to erosion until a complete vegetation cover has been recovered, could also be slowing down the restoration process.

Although no changes in total carbon pools were observed following grazing exclusion, higher litter amount registered in non-grazed stands (Table 3, Fig. 2) could improve soil properties, enhancing rangeland productivity in the long term. A build-up of litter on the

soil surface has proved to be one of the first outcomes observed after livestock removal (Soriano et al., 1980; Reeder and Schuman, 2002). Changes in water dynamics could stem from the higher litter amount. Direct soil evaporation represent in Patagonia more than 50% of the total water losses (Paruelo and Sala, 1995). An increase of litter would reduce this water flux (Soriano et al., 1980) increasing soil water availability in the first soil layers. Shallow rooted plants, e.g. forbs and grasses, which compete actively with direct soil evaporation (Westoby, 1980), would probably be favoured in this process. Higher forb contribution observed in non-grazed stands would support this. Seed retention and seedling establishment could also be enhanced by litter because it generates microhabitats propitious for these processes (Alippe and Soriano, 1978). In addition, higher litter amount on the soil surface is expected to reduce soil temperature, which may in turn decrease soil organic matter decomposition rates and diminish nutrient leaching to deep soil layers (Austin et al., 2004).

Since sheep herds are highly selective and generally prefer grasses rather than shrubs (Golluscio et al., 1998), changes in the grass/shrub ratio are expected after grazing exclusion (Leon and Aguiar, 1985), however our results did not support this expectation (Table 3). One possible cause of this pattern is that shrubs could also be negatively affected by grazing. It has been noted in Patagonia that sheep eat flowers and fruits of the shrub *Mulinum spinosum* hindering the recruitment of new seedlings (Fernández et al., 1992). Lower density and basal cover of this shrub has been observed as grazing intensity increases (Fernández et al., 1992). On the other hand, shrubs are long-lived organisms and once established they may persist regarding the grazing regime, increasing the inertia of the system.

Our results showed that afforestation could yield significant carbon gains in semi-arid rangelands; however environmental consequences, which could emerge from this deep land use shift, must be taken into account. Trees, through their deep root system (Schenk and Jackson, 2002) and high aerodynamic conductance (Kelliher et al., 1993), are able to evapo-transpirate more than herbaceous vegetation (Nosetto et al., 2005). These evapo-transpiration contrasts often translate into lower water yield and ground-water recharge (Jobbágy and Jackson, 2004) which could eventually jeopardize other water uses. Although, the effects of trees on water budget are expected to attenuate towards arid environments because most of the water reaching the soil is evapo-transpired regardless of the vegetation cover (Wilcox, 2003), all those conditions that promote the availability of subsurface water (e.g. decoupling of precipitation and evapo-transpiration, coarse fractured soils, bypass flow) may enhance the effects of this land use shift (Huxman et al., 2005). In addition to changes in water quantity, afforestation could also modify water and soil chemistry through changes in nutrient circulation (Jobbágy and Jackson, 2004). Loss of local biodiversity is another negative outcome of afforestation that must be considered (Peterken, 2001). Identifying and applying management options (e.g. adequate planting distances, thinning, selection of tree species, location of afforested stands) to favor productivity, and consequently carbon gains, minimizing at the same time potential negative effects is a critical challenge in semi-arid rangeland afforestation.

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