

Are there any trade-offs between forage provision and the ecosystem service of C and N storage in arid rangelands?



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ABSTRACT

Changes in land use often increase the provisioning ecosystem services at the cost of decreasing the regulating services. Thus, the appropriation of primary production to optimize the supply of forage for livestock production may undermine C and N storage, essential to maintain the integrity of ecological systems and the biosphere. The aim of this work was to study this trade-off by estimating the effect of grazing intensity on two provisioning and regulating ecosystem services (forage supply and C and N storage, respectively) in a 300 km² Patagonian steppe. In areas with different historical sheep grazing regime (intensive, moderate and ungrazed), we estimated forage supply through the aboveground biomass of preferentially consumed species as well as total C and N storage in plants, through forage and non-forage aboveground biomass, litter and root biomass in the top 20 cm of soil. We found that forage supply and C and N storage were highest in moderately grazed areas and were positively correlated, indicating the absence of trade-offs between them. Grazing exclusion had no effect on total plant C and N, but decreased these stocks in green grass biomass in relation to moderate grazing. Intensive grazing decreased both provisioning and regulating services, markedly diminishing grass C and N stocks and grass forage compared to other conditions. Conversely, shrubs and roots were not influenced by grazing regime. This study provides evidence that in arid rangelands, an adequate grazing management could be a key control to complementarily maximize both provisioning and regulating ecosystem services.

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1. Introduction

Change in land use is one of the major factors affecting terrestrial ecosystem structure (e.g., species composition, C and N stocks), functioning (i.e., community dynamics, primary productivity, decomposition), and ecological services provision (Sala et al., 2000; Millennium Ecosystem Assessment, 2005). The main environmental challenge is to sustain the capacity of ecosystems to provide goods and services meeting current and future human needs (DeFries et al., 2004; Millennium Ecosystem Assessment, 2005; Foley et al., 2005). Several authors stressed that in many ecosystems, trade-offs between regulation and provision ecosystem services are inevitable (DeFries et al., 2004; Millennium Ecosystem Assessment, 2005; Foley et al., 2005; Rodriguez et al., 2006; Bennett et al., 2009; Power, 2010; Raudsepp-Hearne et al., 2010). For example, increasing crop production (provisioning

service) occurs at the expense of losses of regulation services such as carbon sequestration capacity and/or water quality regulation (Foley et al., 2005). In general, in rangeland ecosystems those trade-offs have not been thoroughly evaluated (but see Sala and Paruelo, 1997; Havstad et al., 2007) despite the generalized idea that domestic grazing promotes degradation and desertification (Ares et al., 2003; Reynolds et al., 2007).

In rangelands, the main provisioning service is forage supply for livestock production (kg dry matter ha⁻¹ yr⁻¹) and the main regulating service is C sequestration and storage (kg C ha⁻¹) (Havstad et al., 2007; Yahdjian et al., 2015). Forage provision is the fraction of aboveground biomass that can be consumed by domestic herbivores, which in arid rangelands represents a small fraction of primary production (Golluscio et al., 1998; Oesterheld et al., 1999). In these lands, animal husbandry is an important activity in terms of cultural heritage, and grazing management reduces social impacts in comparison with other land uses (e.g., crop production, afforestation) (Havstad et al., 2007). Furthermore, rangelands represent a vast storage of C, both in soils and vegetation, containing 20–25% of the global terrestrial C

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(Havstad et al., 2007). Both services (forage provision and C sequestration) have a clear global and local impact on human societies and have been the target of many policy interventions in rangelands management; however, the existence of trade-offs between them remain poorly studied. Here we estimated forage provision and C and N sequestration under different grazing management alternatives in arid rangelands. In such areas, trade-offs between these services may occur, for example, when native vegetation is removed to plant pastures in order to optimize forage supply for livestock production. In general, this practice undermines C and N storage (FAO, 2010). On the other hand, grazing can change plant functional type composition, decreasing forage species and promoting a shrub encroachment which may maintain or increase C storage in non-forage shrubs (e.g., Van Auken, 2000) and thus generates the trade-off.

A singularity of arid rangelands is the relative contribution of vegetation and soil to total C storage. Even though soil is the main stock for C sequestration in rangelands (Lal, 2004; FAO, 2010), interestingly, as aridity increases C and N stocks in plant biomass and litter increase in importance compared to soil stocks. Carrera and Bertiller (2010) found that both plant cover and soil C and N stocks decrease along a gradient of increasing aridity, whereas the amount of aboveground litter does not change and woody cover with more recalcitrant material increases. Furthermore, decomposition rates of aboveground vegetation in arid temperate ecosystems dominated by woody species and perennial grasses, is usually slower than in mesic systems, and therefore the longevity of biomass stocks could be higher (Meentemeyer, 1978). On the other hand, the magnitude and importance of grazing effects on soil C and N stocks are elusive because of the direct and indirect effects of grazers, although such effect occurs mostly under heavy grazing pressure (Golluscio et al., 2009). There is also agreement that grazing reduces N availability (e.g., Golluscio et al., 2009). Nitrogen is a restriction in almost all water limited ecosystems (Hooper and Johnson, 1999; Yahdjian et al., 2011). Even though the ecosystem service associated with N per se is nutrient cycling, N availability and N stocks in plants is one of the key aspects for C sequestration (Piñeiro et al., 2010) and for quantity and quality of primary production and forage (West and Skujins, 1978; Burke et al., 1997). Herbivory may alter N cycling by selectively removing biomass, by physical disturbance and by excreting nutrient in feces and urine (Hobbs, 1996, 2006). In addition, herbivory may change plant litter quantity and quality through changes in species composition (Semmartin et al., 2004). In arid steppes, species promoted by grazing contain lower N levels than those diminished by this practice (Semmartin et al., 2004; Vivanco and Austin, 2006), which may involve N depletion in grazed fields.

Our objective was to study sheep grazing effects on (1) above and belowground C and N stocks in plants (regulating service), (2) forage biomass (provisioning service), and (3) the existence of trade-offs between both types of services in a mixed grass and shrub steppe. We specifically estimated forage fraction (within total plant biomass) under three grazing intensities in order to quantify the key provisioning service for livelihood of peasants and ranchers. The general hypothesis was that intensification of domestic grazing reduces the stock sizes of C and N and forage biomass due to selective defoliation. However, if grazing intensity is moderate, this reduction could be compensated because grazing can promote an optimization process (McNaughton, 1979), increasing productivity without a major reduction of more quality forage species, maintaining both types of ecosystem services provision and reducing trade-offs. Because of this optimization process, absence of domestic grazing will not necessarily increase forage, C and N stocks.

2. Materials and methods

2.1. Study site description

We worked in a grass-shrub steppe area of 300 km², including the Rio Mayo INTA Experimental Station and privately owned neighboring rangelands, in South Central Patagonia, Chubut province, Argentina (45° 24' lat. S and 70° 15' long. W). These ecosystems are devoted to wool production and have been grazed by sheep for more than one hundred years. Grazing management is extensive, arranged in very large paddocks (frequently around 2000–5000 ha), which are in general continuously grazed (Golluscio et al., 1998). Few dominant perennial grasses and shrubs contribute approximately 96% of the total biomass (Fernández et al., 1991) and mean aboveground net primary production is 56 g m⁻² y⁻¹, half corresponding to grasses and half to shrubs (Jobbágy and Sala, 2000). Sheep and native herbivores are very selective and select their diet from the dominant grass and shrub species (Aguar and Román, 2007).

2.2. Grazing treatments

We investigated three grazing managements: moderate and intensive grazing, both of which are continuous, and ungrazed (exclosure >20 years). Each management was represented by three replicates (different paddocks or fields). The moderately grazed situations are paddocks from the Experimental Station above mentioned, with a stocking rate of ~0.2 sheep ha⁻¹, during the last 20 years. The intensively grazed situations correspond to paddocks where the stocking rate during the last 20 years was ~0.4 sheep ha⁻¹ (Cipriotti and Aguilar, 2005). In grazed fields we avoided areas near watering or fencing where sheep usually overgraze, making them unrepresentative of the average grazing intensity. The ungrazed condition was evaluated in three exclosures (installed in 1954, 1972, and 1983), for shrub biomass and belowground biomass estimation. Conversely, for destructive sampling needed to estimate grass aboveground biomass we only used the 1983 exclosure which was larger (>5 ha) than the rest, and included enough heterogeneity to reduce pseudoreplication effects. Furthermore, we confirmed that the three exclosures did not differ significantly in shrub and grass species densities. All study sites corresponded to the same plant community. Therefore, differences in species composition among treatments can be attributed to grazing historical effects (Cipriotti and Aguilar, 2005). The study was performed at the end of the growing season (peak of green biomass) during which annual rainfall was similar to an average year (i.e., 156 mm).

2.3. Estimation of regulating and provisioning services

We estimated the regulating service through the capacity of plants to sequester C and N in biomass. To estimate C and N concentrations we harvested and analyzed: (1) all aboveground green and standing dead biomass that was mostly explained by the dominant grass species (*Pappostipa speciosa* Trin. et Rupr., *Pappostipa humilis* Cav., *Poa ligularis* Nees ap. Steud and *Bromus pictus* Hook) and dominant shrub species (*Mulinum spinosum* (Cav.) Pers, *Adesmia volckmannii* Philippi and *Senecio filaginoides* De Candolle), (2) litter, and (3) roots in the top 20 cm of soil from three grazing intensities ($n = 3$). In the case of shrubs, we estimated C and N for main tissues: stem, lateral branches, and terminal twigs including leaves. Samples were homogenized and grounded before analysis. Elemental analyzer for C and N LECO TruSpec CN (St. Joseph, USA, 2004) was used. We estimated plants C and N stocks multiplying concentrations of C and N in each category of biomass (Table 1 in Appendix A) by specific aboveground green

biomass, specific aboveground standing dead biomass, belowground biomass and litter through weighted average. Specific aboveground grass, shrub and litter biomass data (kg ha^{-1}) was taken from Oñatibia and Aguiar (submitted). Belowground biomass was estimated in the three grazing conditions ($n=3$). We located one 50 m transect in each paddock and extracted a core (soil and roots) every two meters using a cylinder (0.06 m diameter and 0.20 m depth). In this shrub-grass steppe, grass roots explore just the top 20–30 cm of the soil profile, while shrub roots explore deeper portions (Fernández and Paruelo, 1988; Sala et al., 1989). We ruled out deeper root biomass of shrubs due to its difficult estimation. Soil sandy texture allowed separation of biomass because the soil particles are easily detached from the roots. The material obtained was dried in an oven (65°C for 72 h) and the resulting dry biomass was weighed (Leva et al., 2009).

We estimated the provisioning service through forage biomass availability to sheep which combined both green biomass estimates and an aptitude index. In this arid ecosystem, forage biomass is a small fraction of total biomass (Golluscio et al., 1998). First we calculated the green biomass of dominant grass and shrub species potentially available for herbivores in each grazing condition. In grasses, we assumed that only green biomass may potentially be forage. In shrub species we applied a coefficient from total green biomass, to estimate the biomass of twigs, leaves and flowers (Oñatibia et al., 2010) in order to discard woody tissue that is not browsed. Green biomass in peak production may be considered as annual productivity in these sites presenting a brief and pronounced growing season (Sala and Austin, 2000). Once the available specific green biomass was obtained, we estimated forage biomass by multiplying species green biomass and forage aptitude factor of each species following Easdale and Aguiar (2012). Also, we estimated average forage consumption by sheep in moderately and intensively grazed condition by multiplying the sheep per ha by the estimated consumption per sheep and year (365 kg of dry matter; Agricultural Research Council, 1980). In the graph, we presented forage results without adding these estimates of sheep consumption due to the simplicity of estimation.

2.4. Data analysis

We analyzed the effects of grazing on C and N stocks with ANOVA (analysis of variance), comparing C and N in aboveground

green, standing dead and total biomass for each life form (grasses and shrubs) as well as in litter and belowground biomass among grazing levels. To evaluate the effect of grazing on forage biomass, we performed an ANOVA among grazing levels comparing total forage biomass of each life form. We also performed two-way ANOVAs with grazing intensity and species to compare forage biomass. Finally, to assess whether there are trade-offs between regulation and provision services, we performed Pearson's correlations between, both, C and N mass and forage biomass in grasses and shrubs of all sites ($n=18$; 3 grazing treatments \times 3 replications \times 2 functional groups).

Analyses were performed using Infostat software (2008). We controlled data normality and variance homogeneity. We used Tukey test for mean comparisons post-hoc ANOVA. Values in the text and graphs are mean \pm standard error. Results of C and N stocks and forage biomass were presented separating grass and shrub categories because of the different estimation approaches (Oñatibia and Aguiar, submitted).

3. Results

3.1. Regulating services: C and N storage in plants

Carbon stock in total aboveground grass biomass was lower in intensively grazed sites ($1034 \pm 110 \text{ kg ha}^{-1}$) compared to ungrazed ($1598 \pm 10 \text{ kg ha}^{-1}$) and moderately grazed ones ($2066 \pm 154 \text{ kg ha}^{-1}$) ($p < 0.05$) (Fig. 1a). In contrast, C stocks of shrubs did not change ($p=0.89$) among grazing conditions (average overall grazing conditions $2361 \pm 109 \text{ kg ha}^{-1}$). Carbon in green biomass of grasses was higher in moderately grazed areas than in intensively grazed and ungrazed ones ($p < 0.05$, Fig. 1a). Standing dead C of grasses was lower in intensively grazed areas than in enclosures or moderately grazed fields ($p < 0.05$, Fig. 1a). In shrubs, C in green and standing dead biomass was not affected by grazing ($p > 0.05$, Fig. 1a), although values of C stock in shrub's green biomass were highly variable under intensive grazing (Fig. 1a). Carbon in litter was higher in moderately grazed areas ($229 \pm 13 \text{ kg ha}^{-1}$) and enclosures ($251 \pm 14 \text{ kg ha}^{-1}$) than in intensively grazed sites ($109 \pm 12 \text{ kg ha}^{-1}$) ($p < 0.05$, Fig. 1a). The C stocks in roots from the top 20 cm of soil did not change with grazing regime ($p=0.40$; $391 \pm 49 \text{ kg ha}^{-1}$).

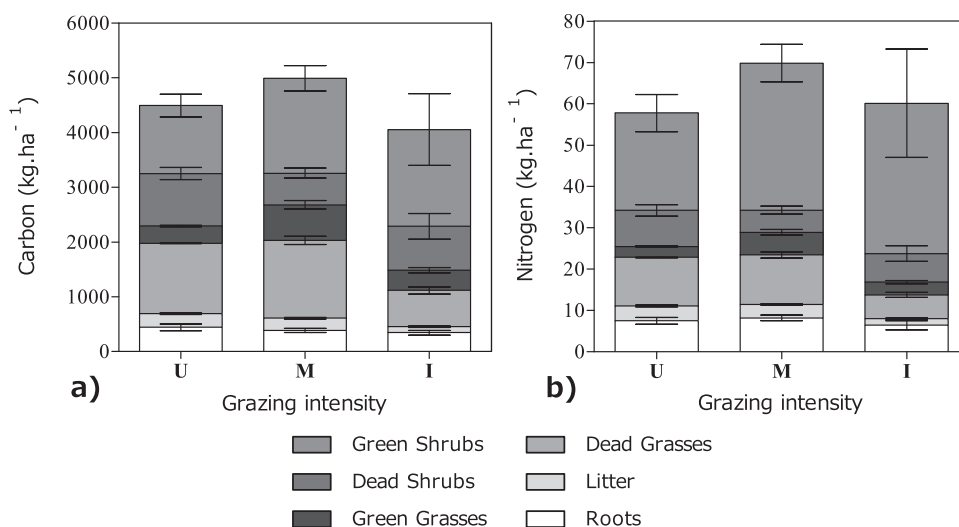


Fig. 1. (a) Carbon and (b) nitrogen stocks in aboveground green biomass of grasses and shrubs, aboveground standing dead biomass of grasses and shrubs, litter and belowground biomass of the top 20 cm of soil under three grazing intensities (U, ungrazed; M, moderately grazed; I, intensively grazed). Bars correspond to mean values and vertical lines indicate standard errors. See the text in Section 3 for pattern description and statistical significances.

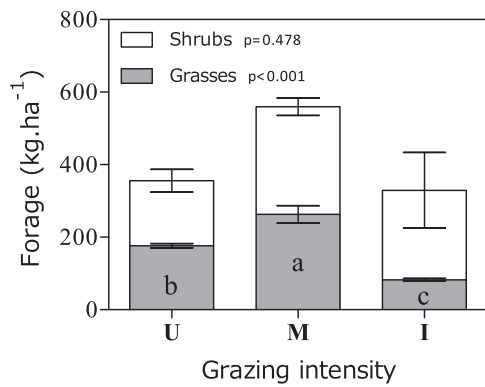


Fig. 2. Forage biomass of grasses and shrubs under three grazing intensities (U, ungrazed; M, moderately grazed; I, intensively grazed). Bars correspond to mean values and vertical lines indicate standard errors. Tables indicate p -values resulting from ANOVA among grazing conditions for each life form. Different letters indicate significant differences ($p < 0.05$) among grazing intensities (Tukey test).

Nitrogen stock in aboveground biomass followed the same pattern as C in all studied pools and grazing conditions (Fig. 1b). Nitrogen in green biomass of grasses was higher ($5.5 \pm 0.7 \text{ kg ha}^{-1}$) in moderately grazed sites ($p < 0.05$) than in ungrazed and intensively grazed areas ($2.7 \pm 0.1 \text{ kg ha}^{-1}$ and $3.1 \pm 0.4 \text{ kg ha}^{-1}$, respectively, Fig. 1b). Nitrogen stock in grass standing dead biomass was lower in intensively grazed areas ($5.7 \pm 0.6 \text{ kg ha}^{-1}$) than in moderately grazed sites ($12 \pm 0.8 \text{ kg ha}^{-1}$) and enclosures ($11.8 \pm 0.1 \text{ kg ha}^{-1}$) paddocks ($p < 0.05$, Fig. 1b). Nitrogen stocks in shrubs did not change with grazing ($p > 0.05$, Fig. 1b). The average values of the three grazing conditions were $31.8 \pm 7.3 \text{ kg ha}^{-1}$ in green biomass and $7 \pm 1.4 \text{ kg ha}^{-1}$ in standing dead biomass, being shrub green biomass the main N pool. Nitrogen stock in litter was higher ($p < 0.05$) in moderate grazing ($3.3 \pm 0.2 \text{ kg ha}^{-1}$) and enclosures ($3.6 \pm 0.2 \text{ kg ha}^{-1}$) than in intensive grazing ($1.6 \pm 0.2 \text{ kg ha}^{-1}$). Root N stock was low ($7.4 \pm 0.5 \text{ kg ha}^{-1}$) and did not change with grazing management ($p = 0.43$; Fig. 1b).

3.2. Provisioning service: forage for sheep

Grass forage biomass was modified by grazing management ($p < 0.05$) and presented maximum values in moderately grazed areas ($263 \pm 24.1 \text{ kg ha}^{-1}$), intermediate values in ungrazed ones ($176.3 \pm 6.2 \text{ kg ha}^{-1}$) and minimum values in intensively grazed sites ($82.6 \pm 3.4 \text{ kg ha}^{-1}$) (Fig. 2). Grazing management did not change shrub forage biomass ($p = 0.48$) (overall mean: $241 \pm 36.4 \text{ kg ha}^{-1}$; Fig. 2). Patterns of total forage biomass were explained by the different species response to grazing (Table 2 in Appendix A). Forage of *Adesmia volckmannii* and *Pappostipa speciosa* were higher in moderate grazing than other grazing conditions. *Poa ligularis* showed a tendency to decrease and *Mulinum spinosum* to increase with grazing (Table 2 in Appendix A). Average forage consumed by sheep was 73 and 146 kg ha^{-1} in moderately and intensively grazed paddocks, respectively.

3.3. Trade-offs

C and N stocks in grasses and shrubs were positively correlated with forage biomass of grasses and shrubs (Fig. 3). The correlation coefficient between forage biomass and C mass (Pearson = 0.90, $R^2 = 0.81$, $p < 0.001$) was higher than with N mass (Pearson = 0.79, $R^2 = 0.64$, $p < 0.001$). All points of moderately grazed sites were above the correlation lines, indicating that forage biomass/C stocks and forage biomass/N stocks ratios were higher than under other grazing situations. On the other hand, shrubs values under

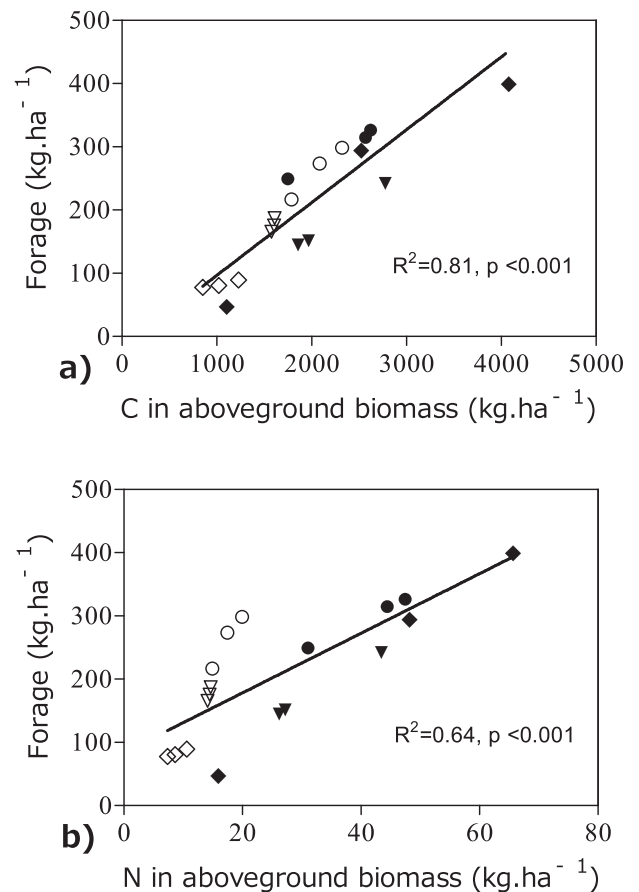


Fig. 3. Correlations between (a) carbon and (b) nitrogen in grasses and shrubs (stock) and forage biomass (forage supply) in the same two life forms under different grazing intensities (white triangles represent grasses under enclosure, black triangles represent shrubs under enclosure, white circles represent grasses under moderate grazing, black circles represent shrubs under moderate grazing, white rhombus represent grasses under intense grazing, black rhombus represent shrubs under intense grazing).

intensive grazing had high variability and displayed the whole range of values in both correlations (Fig. 3).

4. Discussion

Here we showed that both regulating and provisioning services studied were, in general, maximized at moderate grazing intensity. Moreover, they were positively correlated, indicating the absence of trade-offs between them. This absence occurs because grazing modified productivity and C sequestration without producing major changes in species composition. Intensification of grazing could eventually reduce forage biomass and decrease forage/C storage ratio, because of disappearance of preferred species. However, in this study, forage biomass roughly followed the pattern of C storage. Our results would not be drastically changed by considering the forage consumed by sheep in intensively grazed paddocks. Strikingly, both no grazing and intensive grazing reduced C and N stocks (10% and 20%, respectively) compared with moderately grazed areas. These results support the notion that management of natural vegetation through grazing with moderate sheep stocking rate would be an effective tool to maximize regulating and provisioning ecological services in these arid rangelands. Being domestic livestock the most common and widespread land use in Patagonian rangelands and elsewhere, our results open a new perspective for analysis of rural socio-ecosystems.

Grass stocks were more sensitive than shrub stocks to grazing management. In intensively grazed fields the amounts of aboveground grass C decreased by almost half when compared with the other two conditions. Recovering this stock requires more than seven years of biomass accumulation (regardless of decomposition) according to aboveground grass production estimated by Jobbágy and Sala (2000) in ungrazed areas. These results suggest that intensive grazing reduces the natural capital which is the basis of the sheep industry in Patagonia (Ares, 2007). Then, increasing the biomass of herbivores in an attempt to achieve a greater secondary productivity per area unit (mainly kg wool ha⁻¹ yr⁻¹, but also kg meat ha⁻¹ yr⁻¹) would not be sustainable because it reduces the amount of forage biomass diminishing future secondary production (Ares, 2007).

The exclusion of grazing did not change total C and N stocks of grasses, but decreased green biomass stocks and forage biomass in relation to moderate grazing. Grazing can have positive effects on aboveground productivity, as demonstrated for some grassland and savanna ecosystems (Oesterheld et al., 1999). It has been suggested that moderate grazing can promote an optimization process of production (McNaughton, 1979). Under these conditions, productivity would increase by compensatory growth of plants (McNaughton, 1983), which would increase the green biomass and revitalize the system, despite the biomass removal by animals. Milchunas and Lauenroth (1993), found that grazing slightly increases ANPP in some communities where grazing is moderate, the ANPP is low and the evolutionary history is long. These conditions may be found in these Patagonian steppes (Lauenroth, 1998). We suggest that, in this rangeland, exclusion of grazing would not be an appropriate management practice to increase the sequestration of C and N and forage biomass.

Shrub C and N stocks and shrub forage biomass were, on average, not affected by the range of sheep grazing management studied. Nevertheless, under intensive grazing, C and N stocks were highly variable, decreasing or increasing depending on the paddock measured. There has been an intense debate about the woody species encroachment in arid and semi-arid steppes (Sankaran et al., 2004). Our results may be indicative that grazing intensification in this steppe, could trigger different and opposite responses, from shrub encroachment to shrub decrease (Cipriotti and Aguiar, 2012).

Changes induced by grazing in soil C stocks reported for this steppe (Golluscio et al., 2009) are minor compared to aboveground stock changes. The reduction in grass aboveground C stocks under intensive grazing was significant and greater than the reduction measured in soil C in this same steppe (ca. 1000 kg ha⁻¹ vs 616 kg ha⁻¹, respectively; soil C estimation from Golluscio et al., 2009). The lack of significance in soil organic matter results also indicates that soil stabilization service would not be markedly affected by grazing. Green biomass, which can be considered a labile C stock, explained most of the difference in C storage. However, more recalcitrant stocks such as wood, dead material, litter and roots did not differ between moderately grazed and ungrazed areas, but they were higher than in intensively grazed paddocks.

The total amount of N in grasses and shrubs followed C patterns but implications may be different. Nitrogen stocks in shrubs and roots were not modified by grazing. However, N stocks in grasses decreased under intensively grazed conditions, mainly due to biomass reduction. The change in biomass species composition caused by grazing (Oñatibia and Aguiar, submitted.) had no marked effect on total amount of N in aboveground biomass of vegetation. This is because, on the one hand, the quality of biomass in terms of N concentration did not vary considerably between decreasing and increasing species (Table 1 in Appendix A). In addition, grazing did not alter N concentration in most species (Table 1 in Appendix A). On the other hand, biomass loss of species with higher N

concentrations was compensated (partially or totally, depending on grazing intensity) by an increase in the biomass of species with lower N concentration (Oñatibia and Aguiar, submitted.). These results partially support the hypothesis that grazing has a negative effect on nutrient stocks causing desertification due to fertility loss (Golluscio, 2002) since N stock in rangeland vegetation is only reduced if the stocking rate management is inadequate. Finally, the increase of species with lower N content and lower decomposition rate in grazed conditions (e.g., *Pappostipa speciosa*) (Semmartin et al., 2004; Vivanco and Austin, 2006; Oñatibia and Aguiar, submitted), would raise the average longevity of biomass and therefore it would increase the residence time of C with positive implications for C sequestration in aboveground biomass as a sink.

Forage production was maximized at moderate grazing intensity although forage biomass was estimated after sheep consumption. If we include this forage, our estimation rises considerably in both moderately and intensively grazed sites. Anyway, including sheep consumption, forage production during the study year was over 20% higher in moderately grazed areas than in exclosures and intensively grazed ones. In these steppes common agricultural practice to increase primary production (replacement of natural vegetation for improved grass species, fertilization) incurs in energy and capital subsidies at high failure risk because of temporal and spatial variability of environmental conditions (Easdale and Aguiar, 2012). We claim that in these grass-shrub steppes moderate grazing use is the best option for land use (De Pauw et al., 2000). Previous studies in this steppe support the idea that another valuable ecosystem service such as biodiversity (Noss, 1990) is not negatively affected by moderate grazing. Plant richness and diversity (compositional component, Perelman et al., 1997) as well as the spatial organization of the vegetation mosaic (structural component, Cipriotti and Aguiar, 2005) did not change with moderate domestic grazing. Our results may be valid for other grazing ecosystems with biophysical constraints (poor soils, water availability shortage during most of the year, high insolation, and constant dry winds). Indeed, domestic grazing could complementary promote both types of services, and therefore it would have a direct positive impact on the socio-economic viability of rural production systems, reducing socio-economic cost of rural to urban migration (Aguiar and Román, 2007).

We propose that although changes in aboveground stocks per unit of area were not very high in absolute terms with respect to other ecosystem types, they become important when considering the total area covered by arid rangelands (25% of Earth's land surface, Asner et al., 2004). For example, taking into account the area covered by the Occidental District of the Patagonian steppe (90,000 km², Soriano et al., 1983), the quantity of C could increase by nine million tons only considering plant aboveground biomass of this region, exclusively through grazing management. Furthermore, because forage productivity is also maximized, animal farming industry would be benefited reducing the people migration to urban centers (Aguiar and Román, 2007). Additionally, the viability of afforestation or other vegetation changes, as well as being constrained by abiotic conditions (i.e., water and temperature stresses), is strongly controlled by investment in energy subsidies. In this regard, assertions stating that grazing is the worst option for the use of natural resources in Patagonia arid rangelands (Murdoch et al., 2010) appear as requiring at least further critical evaluation in the light of specific data for the system in questions, such as the one contributed in this work.

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Appendix A. Complementary data

Table 1

C and N concentration (%) in aboveground green and standing dead biomass of dominant species (*Av*, *Adesmia volckmannii*; *Ms*, *Mulinum spinosum*; *Sf*, *Senecio filaginoides*; *Ps*, *Pappostipa speciosa*; *Ph*, *Pappostipa humilis*; *Pl*, *Poa ligularis*; *Bp*, *Bromus pictus*), litter and roots biomass under three grazing intensities.

Life form	Species	Biomass category	Grazing intensity					
			Ungrazed		Moderately grazed		Intensively grazed	
			Carbon	Nitrogen	Carbon	Nitrogen	Carbon	Nitrogen
Grasses	<i>Ps</i>	Green	46.87 ± 0.32	0.34 ± 0.03	46.47 ± 0.85	0.37 ± 0.06	47.50 ± 0.20	0.39 ± 0.02
		Standing dead	45.10 ± 0.36	0.38 ± 0.02	45.20 ± 0.53	0.36 ± 0.02	44.97 ± 0.47	0.38 ± 0.05
	<i>Ph</i>	Green	47.10 ± 0.36	0.43 ± 0.01	47.40 ± 0.00	0.47 ± 0.07	47.63 ± 0.35	0.40 ± 0.06
		Standing dead	45.67 ± 0.67	0.44 ± 0.03	46.20 ± 1.05	0.43 ± 0.05	45.67 ± 0.42	0.40 ± 0.04
	<i>Pl</i>	Green	45.30 ± 0.79	0.37 ± 0.04	45.00 ± 0.26	0.37 ± 0.06	43.77 ± 0.23	0.42 ± 0.04
		Standing dead	43.77 ± 0.51	0.40 ± 0.03	43.10 ± 0.26	0.41 ± 0.06	41.40 ± 0.53	0.41 ± 0.05
	<i>Bp</i>	Green	41.27 ± 0.55	0.49 ± 0.01	42.50 ± 0.20	0.65 ± 0.02	41.73 ± 0.46	0.55 ± 0.03
		Standing dead	38.73 ± 0.15	0.42 ± 0.03	40.53 ± 0.35	0.49 ± 0.04	41.03 ± 0.75	0.54 ± 0.02
Shrubs	<i>Av</i>	Green	50.20 ± 0.30	1.16 ± 0.02	49.90 ± 0.20	1.26 ± 0.06	50.30 ± 0.36	1.32 ± 0.14
		Standing dead	54.07 ± 0.74	0.67 ± 0.09	54.57 ± 0.29	0.63 ± 0.09	53.73 ± 0.71	0.62 ± 0.08
	<i>Ms</i>	Green	50.37 ± 0.32	0.83 ± 0.10	49.97 ± 0.25	0.85 ± 0.09	49.83 ± 0.21	0.87 ± 0.01
		Standing dead	51.10 ± 0.20	0.36 ± 0.01	50.87 ± 0.21	0.43 ± 0.08	50.47 ± 0.15	0.40 ± 0.08
	<i>Sf</i>	Green	50.20 ± 1.04	0.79 ± 0.15	49.97 ± 0.25	0.90 ± 0.08	49.33 ± 0.29	0.99 ± 0.03
		Standing dead	51.57 ± 0.21	0.59 ± 0.04	51.20 ± 0.10	0.51 ± 0.02	50.93 ± 0.29	0.44 ± 0.05
	Litter		41.07 ± 1.27	0.64 ± 0.06	39.90 ± 1.87	0.58 ± 0.10	42.43 ± 0.55	0.54 ± 0.14
	Roots		34.42 ± 0.90	0.59 ± 0.01	32.05 ± 0.82	0.68 ± 0.01	34.65 ± 1.06	0.64 ± 0.01

Table 2

Effect of grazing intensity on forage biomass obtained for each condition considering potentially forage green biomass in peak production and specific forage aptitude (Easdale and Aguiar, 2012). Different letters indicate significant differences in forage biomass ($p < 0.05$) product of a two way ANOVA between grazing intensity and species (Tukey test).

Grazing intensity	Species	Potentially forage green biomass (kg ha ⁻¹)	Forage aptitude	Forage biomass (kg ha ⁻¹)
Ungrazed	<i>Pappostipa speciosa</i>	38.27	0.5	19.13 ab
	<i>Pappostipa humilis</i>	34.97	0	0 a
	<i>Poa ligularis</i>	140.4	1	140.4 bc
	<i>Bromus pictus</i>	16.75	1	16.75 ab
	<i>Adesmia volckmannii</i>	199.43	0.5	99.72 abc
	<i>Mulinum spinosum</i>	159.6	0.5	79.8 abc
	<i>Senecio filaginoides</i>	135.06	0	0 a
Moderately grazed	<i>Pappostipa speciosa</i>	269.63	0.5	134.81 bc
	<i>Pappostipa humilis</i>	72.34	0	0 a
	<i>Poa ligularis</i>	126.02	1	126.02 abc
	<i>Bromus pictus</i>	2.17	1	2.17 a
	<i>Adesmia volckmannii</i>	284.17	0.5	142.08 bc
	<i>Mulinum spinosum</i>	309.44	0.5	154.72 c
	<i>Senecio filaginoides</i>	101.14	0	0 a
Intensively grazed	<i>Pappostipa speciosa</i>	130.78	0.5	65.39 abc
	<i>Pappostipa humilis</i>	113.93	0	0 a
	<i>Poa ligularis</i>	16.61	1	16.61 ab
	<i>Bromus pictus</i>	0.62	1	0.62 a
	<i>Adesmia volckmannii</i>	185.4	0.5	92.7 abc
	<i>Mulinum spinosum</i>	307.94	0.5	153.97 c
	<i>Senecio filaginoides</i>	216.21	0	0 a

See Tables 1 and 2

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