

## Restoration ability of seasonal exclosures under different woodland degradation stages in semiarid Chaco rangelands of Argentina



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### ABSTRACT

Exclosures are widely used for rangeland restoration in semiarid woodlands. However, grass recovery could be hampered if degradation exceeded certain thresholds. In this study we assessed four years effects of seasonally grazed exclosures –vs. open rangelands– on understory cover (grasses, low shrubs and litter) and peak standing biomass in three increasing degradation stages –mature forests, secondary forests and shrublands– in semiarid Chaco woodlands. We found that grass cover and biomass increased tenfold in four years in mature and secondary forests but remained virtually null in shrublands. In rested forests, the grass cover increments remained relatively constant regardless the annual rainfall amount, both in the driest year 2013 (531 mm) and the wettest year 2015 (924 mm). Only in an extraordinarily wet year (2015) did grass biomass increase in rested shrublands. In that wet year, low shrubs cover –higher at higher overstory degradation– decreased in all woodlands. Our results suggest that shrublands may constitute a new steady state unable to recover by grazing exclusion itself, but probably by its combination with wet periods.

### 1. Introduction

Dry rangelands cover over 39,000,000 km<sup>2</sup> (one quarter of the Earth's land surface) and are inhabited by almost one billion rural people (MAE, 2005). In many cases rangelands are susceptible to degradation driven by overgrazing, and also by selective logging and deforestation, linked to crop expansion, in dry forest and woodlands (Hoekstra et al., 2005). Deforestation, which involves the most drastic change in ecosystems, environment and society (Mustard et al., 2004; Viglizzo and Jobbágy, 2010), also displaces livestock to drier areas, thus increasing pressure on these lands, often unproductive and already degraded after a history of continuous grazing and other anthropic actions (Morello et al., 2012). Hence, addressing rangeland degradation is a complex and challenging priority, because of its ecological, economic and social implications.

Rangeland degradation has been reported in many semiarid areas of South America (Adámoli et al., 1990), North America (Jones, 2000), Africa (Downing, 1978), Asia (Mirzabaev et al., 2016) and Oceania (Yates et al., 2000). Overgrazing reduces grass cover, density and biomass (Yayneshet et al., 2009; Verdoodt et al., 2010), increases bare soil

and promotes shrub encroachment (Van Auken, 2009). These changes in vegetation structure often reduce soil moisture (Branson and Reid, 1981) and nutrient cycling (Golluscio et al., 2009). This mainly affects the upper soil layer (Abril and Bucher, 1999), and thus herbaceous plants with shallow roots.

Until recent decades, grazing was considered the main driver of rangeland dynamics, assumed to be linear and reversible according to the range condition model (Dyksterhuis, 1949). This model predicts that a decrease in grazing intensity will result in secondary succession towards a single equilibrium state or climax that represents the best range condition, driven mainly by biotic (plant-herbivore and plant-plant) interactions. This was questioned for arid and semiarid rangelands since their vegetation and grazing dynamics, strongly driven by stochastic abiotic factors such as remarkably variable rainfall and high spatial heterogeneity, are often discontinuous and irreversible and they may have none or multiple equilibrium states (Ellis and Swift, 1988; Westoby et al., 1989). Accordingly, the state-and-transition model proposes “opportunistic strategies” based on certain environmental conditions and/or management actions (such as grazing rest) to promote system transitions towards more desirable steady states (Westoby

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et al., 1989). Since then, the debate on the relevance of equilibrium and non-equilibrium models had emerged, focusing on the relative weight of biotic and abiotic factors on rangeland dynamics (Illius and O'Connor, 1999; Sullivan and Rohde, 2002). Currently, it is proposed that many semiarid rangelands encompass elements of both equilibrium and non-equilibrium paradigms at different spatial and temporal scales (Briske et al., 2003; Vetter, 2005).

Establishment of exclosures, where livestock grazing is excluded during the growing season, has become an important restoration strategy in semiarid rangelands. Seasonal exclosures have both positive ecological and economic effects, at different temporal scales. In the short term, they provide dry season forage for livestock (Mwilawa et al., 2008) and, in the mid-to long-term, they foster vegetation restoration or rehabilitation (Verdoodt et al., 2010). Rest during the growing season enables flowering and seed production, enriches the soil seed bank (Tessema et al., 2012), increases biomass production (Oba et al., 2000) and litter accumulation (Descheemaeker et al., 2006). As a consequence, soil physical, chemical and biological characteristics improve in the long term (Verdoodt et al., 2009; Raiesi and Riahi, 2014).

Despite the beneficial effects attributed to exclosures, in severely damaged rangelands, herbaceous recovery may be hampered by degradation thresholds, both biotic (e.g. shrub competition; Ratajczak et al., 2014) and abiotic (e.g. water shortage; Holmgren and Scheffer, 2001). Such thresholds are strongly determined by the dominant type of plant cover, particularly in heterogeneous woodlands. It is well known that under-canopy patches store more water in the soil and are more productive than inter-canopy patches (Breshears et al., 1998). Furthermore, the type of dominant coverage determines the quantity and quality of litter, biological activity, soil organic matter and nutrient availability (Abril et al., 1993; Raiesi and Riahi, 2014). However, despite their importance, the influence of the woodland patch type on the effectiveness of the exclosure has been scarcely addressed.

In the semiarid portion of the Dry Chaco ecoregion, continuous grazing, selective logging and charcoal production, turned the natural xerophytic thorny forests (comprised of four layers: upper and lower arboreal, shrubby and herbaceous), savannas and grasslands landscape into a more complex matrix of depleted forests, secondary forests and shrublands, with a widely degraded herbaceous layer. In degraded Chaco rangelands, grasses cover less than 2% and yield less than 100 Kg DM ha<sup>-1</sup> year<sup>-1</sup>, which implies that a potential carrying capacity of about 4–5 ha AU<sup>-1</sup> is currently reduced to 20–40 ha AU<sup>-1</sup> (AU = animal units, equivalent to a cow of 400 kg which breeds a calf per year) (Adámoli et al., 1990; Kunst et al., 2006; Morello et al., 2012). This strongly affects livestock production, the main economic activity of traditional herders who represent more than 80% of livestock producers in the Dry Chaco. Degradation in the Dry Chaco began with the permanent settlement of livestock and forestry in the early twentieth century, but sharply worsened in recent decades with deforestation for crop expansion and livestock intensification (Viglizzo and Jobbágy, 2010; Morello et al., 2012). In Argentina, there is recent legislation that aims to protect native forests and foster multiple sustainable uses. As in other semiarid regions worldwide, seasonal exclosure is one of the increasingly promoted strategies. Nevertheless, exclosures are established indistinctly in different woodland patch types, ranging from less to highly degraded, despite understory conditions significantly differing among them (Abril et al., 1993; Abril and Bucher, 1999).

The main objective of this study was to assess the effects of seasonal exclosures on understory cover and biomass in three increasing degradation stages –mature forests, secondary forests and shrublands– in semiarid Chaco woodlands. Throughout a four year experiment, we monitored structural traits in seasonal exclosures and adjacent rangelands under continuous grazing in these woodlands in four sites in Santiago del Estero (NW Argentina).

## 2. Materials and methods

### 2.1. Study area

Our study was conducted 20 km away from Añatuya city (28°20′ – 28°35′ S latitude; 62°23′ – 62°42′ W longitude), Santiago del Estero province, Argentina. This area corresponds to the semiarid Chaco ecoregion (Morello et al., 2012). Mean annual rainfall is 640 mm (1912–2012) and mean annual hydric deficit is about 300 mm. Annual rainfall during the study period was 656 mm (2012), 531 mm (2013), 779 mm (2014) and 924 mm (2015), coinciding with a high intensity El Niño Southern Oscillation (ENSO) event, which dramatically increases rainfall in the Dry Chaco region. Climate is subtropical, with wet season occurring during the warm period, between October and April, and dry season occurring during the cold period, between May and September. Average temperature in the coldest months is 16.6 °C, while in the warmest months it is 28.8 °C. However, historical absolute maximum temperatures have reached 49.5 °C. Topography is flat, with slopes lower than 0.5%, consisting of dominant plains crossed by ancient river-beds (Peña-Zubiate and Salazar Lea Plaza, 1982). Soils on the plains are fine-textured loamy entic Haplustols, throughout the profile. Soils of ancient river-beds are coarse textured and shallower. The natural vegetation is closely related to the landscape, with forests predominating in the plains, and open grasslands (dominated by *Elionurus muticus*) in the ancient river-beds.

The native forest of the study area is a semi-deciduous xerophytic forest comprised by four layers: upper arboreal, dominated by *Schinopsis lorentzii* (“quebracho colorado”) and *Aspidosperma quebracho blanco* (“quebracho blanco”); lower arboreal, mainly composed by species of the genera *Prosopis* (“algarrobo”); shrubby, consisting of species of the genera *Acacia*, *Atamisquea*, *Celtis*, *Larrea*, *Prosopis* and *Schinus*; and herbaceous, composed by C4 grasses, mostly of the genera *Digitaria*, *Pappophorum*, *Setaria* and *Trichloris*, which coexist with low shrubs of the genera *Capparis*, *Ephedra* and *Celtis*, various forbs of the family *Malvaceae*, *Verbenaceae* and *Acanthaceae* and other *Bromeliaceae* and *Cactaceae* plants. As we mentioned above, natural forest areas have been widely transformed into degraded woodlands with a severely degraded herbaceous layer.

### 2.2. Woodland degradation stages

Our study was carried out only in the plains (covered by xerophilous forests), where at least three woodland types with increasing degradation stages were described (Tálamo and Caziani, 2003; Bonino and Araujo, 2005; Brassiolo, 2005): (1) Mature forest or “bosque de dos quebrachos”, dominated by the upper tree layer, it occupies areas that were selectively logged for timber, tannin and charcoal production, almost a century ago; (2) Secondary forest or “algarrobal”, dominated by the lower tree layer, it develops on areas where agricultural practices, mainly cotton, were abandoned about three decades ago; and (3) Shrubland or “fachinal”, areas where trees are virtually absent, severely degraded by varied combinations of continuous grazing, recurrent anthropic fires, selective logging and crops.

### 2.3. Experimental layout and methods

In order to test the effects of seasonal exclosure and patch type on understory vegetation, we performed a factorial experiment between 2011 and 2015 at four sites (true replicates) located close to Añatuya city. We followed a Randomized Complete Block design (RCB) to assign the main plots ‘grazing management’ (two levels: rest and continuous within each site) and split-plot to the subplots ‘patch type’ (three levels: mature forest, secondary forest and shrubland), within each grazing level.

### 2.3.1. Understorey cover and biomass

At each of four sites, 20 km apart of each other, we identified two adjacent and similar open areas of 10 ha, then we randomly selected one of them and established an enclosure in November 2011. Enclosures were rested during the subsequent growing seasons (from October to April) and they were opened for livestock grazing during the dry season (between May and September). Open areas remained continuously grazed. The selection criteria were the similarity of landscape position (plains), soil type (Añatuya series), current use (continuous grazing, no cropping use in the past three decades), livestock species (cattle and goats), stocking rate (about 5 ha/AU), waterpoint distance (300–600 m) and grass cover (less than 2%). At each enclosure and its adjacent open area, ten homogeneous patches were classified based on the woody component (tree and shrub density and cover) following the methodology proposed by Thren et al. (1993) for semiarid Chaco. Then, three patches of 1000 m<sup>2</sup> were randomly selected among those previously classified, one for each type of woodland: mature forest, secondary forest and shrubland.

Cover of grasses, low shrubs, litter and bare soil was estimated using the “Canopy – Coverage Method” (Daubenmire, 1959). Three 10 m permanent transects per plot were randomly placed in November 2011. Throughout each transect, cover was estimated from 10 metallic frames of 0.2 m × 0.5 m (0.1 m<sup>2</sup>) located at regular intervals, at the end of the growing season (April). Peak standing biomass of grasses and low shrubs were measured by biomass harvest in April, by clipping at 5 cm above the ground level, all standing biomass rooted within an area of 1 m × 10 m (10 m<sup>2</sup>) per plot.

Statistical analysis was performed by repeated measures ANOVA (rmANOVA), with ‘grazing management’ as main effect, and ‘patch type’ and sampling year as within-subject effects, using STATISTICA version 8.0 (StatSoft Inc., 2007) and INFOSTAT (Di Rienzo et al., 2015) statistical software. The correlation between cover and biomass of grasses and low shrubs was analyzed through linear regression analysis using the least squares method, with data from the last three years (2013, 2014 and 2015; 2012 was not considered because cages used to harvest biomass would have avoid grazing, thus producing the same effect as enclosures during the first growing season) and the three patch types of woodland, using software GRAPHPAD PRISM version 5.0 for Windows (GraphPad Software Inc., 2007).

## 3. Results

### 3.1. Understorey cover

Grass cover was higher ( $P < 0.05$ ) in seasonal enclosures than in the adjacent open areas from the second growing season resting both in mature and secondary forests, but it did not differ significantly among treatments in the shrublands (Table 1: F x G and Y x G significant interactions; Fig. 1 A, B, C). Inside the enclosures of both mature and secondary forests, grass cover increased from  $3 \pm 3\%$  to  $28 \pm 5\%$  after four consecutive years of grazing exclusion during wet seasons. Shrublands showed a grass cover of around  $4 \pm 3\%$ , without

significant changes since the enclosures were established. However, a clear upward trend was observed in rested shrublands during the exceptionally wet year 2015 (Fig. 1 C). Despite the fact that low-shrub cover showed responses that are hard to interpret (see triple significant interaction in Table 1: F x Y x G), it appears to be higher under severe overstorey degradation conditions, and lower under the combined conditions of grazing enclosure and the wet period (2015) (Fig. 1 D, E, F). Litter cover was only affected by the year, rising in 2015, and was not affected by grazing or patch type of woodland (Fig. 1 G, H, I). Bare soil proportion was significantly lower inside the enclosures than in the adjacent open areas in secondary and mature forests but not in shrublands (Table 1: G x F significant interaction) (Fig. 1 J, K, L). Furthermore, bare soil tended to decrease throughout the years in mature and secondary forests but remained unchanged in shrublands (Table 1: F x Y significant interaction).

### 3.2. Understorey biomass

Grass biomass showed significant interaction effects among grazing treatments, woodland patch types and years (Table 1: F x Y x G), while low shrub biomass was not affected by grazing enclosure. Grass biomass was higher ( $P < 0.05$ ) inside the enclosures compared to the adjacent open areas during the three monitored years in both mature and secondary forests, but only in the rainy year 2015 in the shrublands. Grass biomass in degraded open areas was about  $80 \pm 70$  Kg DM ha<sup>-1</sup>. Inside enclosures of mature and secondary forests, grass biomass was about  $600 \pm 300$  Kg DM ha<sup>-1</sup>. While secondary forests showed more stability among years, mature forests showed the highest value in a wet year, surpassing  $1050 \pm 450$  Kg DM ha<sup>-1</sup>. Conversely, rested shrublands showed high inter-annual variability: although grass biomass was equal to the open areas during 2013 and 2014, it sharply increased during the extraordinarily wet year 2015, matching biomass of secondary forests. For low shrubs, biomass was  $170 \pm 80$  Kg DM ha<sup>-1</sup> (Fig. 2).

### 3.3. Cover – biomass relationship

We found a significant correlation ( $P < 0.05$ ) between plant cover and biomass for both grasses and low shrubs. As within each functional group this correlation was very similar among woodland types and years, it was well described by an overall correlation (Fig. 3). The slope of the relationship was more than 3 times steeper for grasses than for shrubs.

## 4. Discussion

Grass cover and peak standing biomass (assumed equal to grass productivity; Sala and Austin, 2000) responded positively to grazing rest in the forests -the lowest and intermediate degradation stages- regardless the annual rainfall amount; grass cover increments remained relatively constant both in the driest year 2013 (531 mm) and the wettest 2015 (924 mm). In shrublands -the highest degradation stage-,

**Table 1**

Results from rmANOVA of the effects of grazing management, forest/woodland patch type and year on understorey cover and biomass.

Variable	Grazing (G)		Patch type (F)		F x G		Year (Y)		Y x G		F x Y		F x Y x G	
	F	P	F	P	F	P	F	P	F	P	F	P	F	P
<i>Cover (%)</i>														
Grasses	18.29	0.0052	5.64	0.0188	4.60	0.0330	21.69	< 0.0001	7.56	0.0018	1.01	0.4321	0.81	0.5719
Low shrubs	0.50	0.5026	4.20	0.0414	0.64	0.5465	8.02	0.0013	4.05	0.0231	3.08	0.0153	2.78	0.0252
Litter	0.03	0.9596	1.18	0.3561	0.23	0.8020	6.70	0.0066	0.03	0.9930	0.12	0.9921	1.00	0.4449
Bare soil	34.88	0.0020	39.72	< 0.0001	10.56	0.0034	4.61	0.0177	1.12	0.3716	3.72	0.0070	1.09	0.3885
<i>Biomass (Kg DM ha<sup>-1</sup>)</i>														
Grasses	84.75	< 0.0001	6.24	0.0139	3.76	0.054	17.14	0.0003	8.90	0.0043	3.22	0.0298	3.32	0.0268
Low shrubs	0.02	0.8988	0.38	0.6929	0.02	0.9837	6.88	0.0102	0.03	0.9670	3.76	0.0164	0.04	0.9971

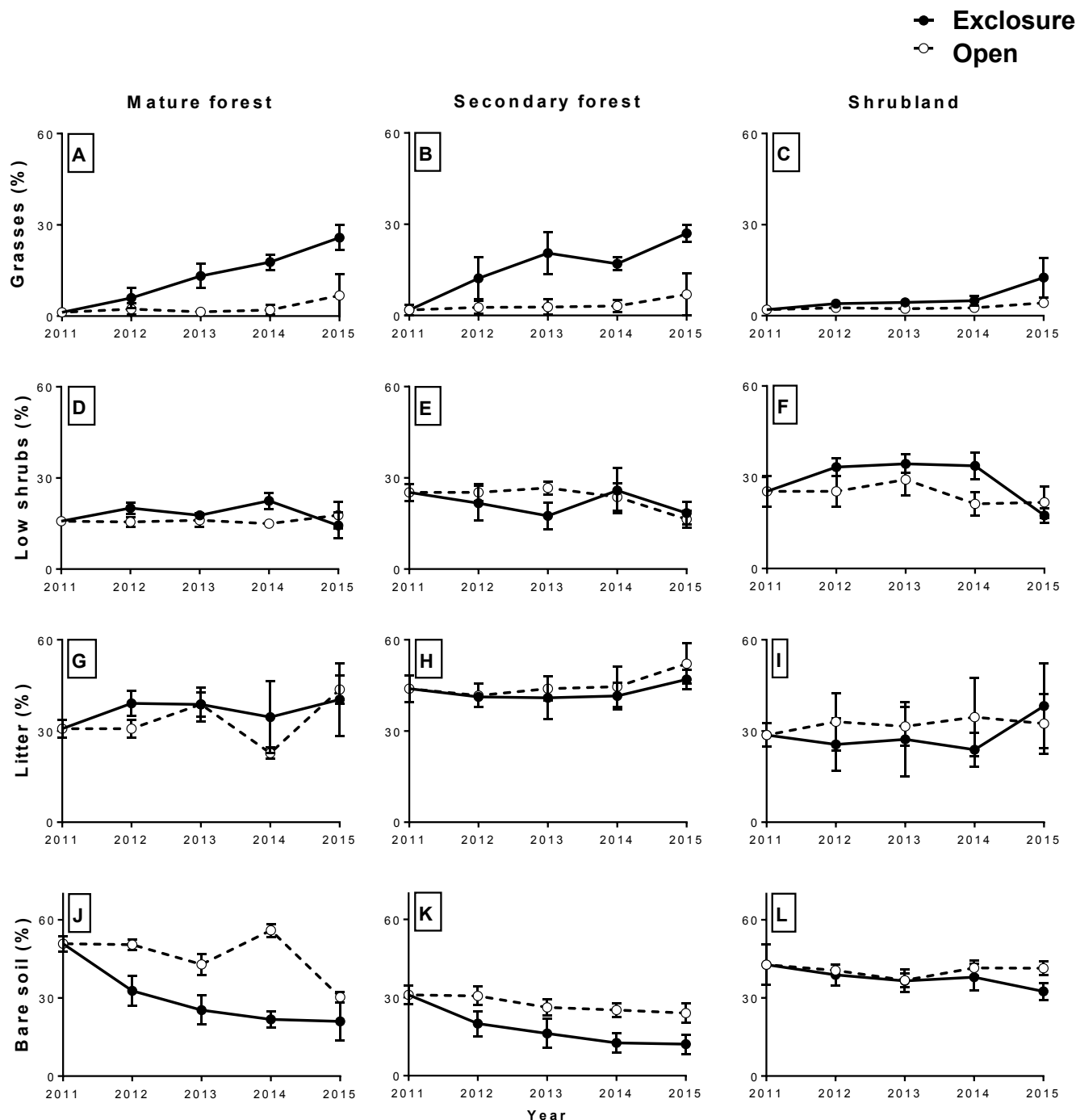


Fig. 1. Cover of grasses (A, B, C), low shrubs (D, E, F), litter (G, H, I) and bare soil (J, K, L) in continuously grazed open areas (open circles, dashed lines) and seasonal exclosures (filled circles, continuous lines) for a period of four-years since the establishment of the exclosures in three patch types of woodland (mature forest, secondary forest and shrubland) of semiarid Chaco. Bars represent standard error mean.

grass biomass showed signs of recovery only under wet conditions, generated by “El Niño” event in 2015, combined with grazing rest allowed by exclosures. In this case, abiotic and biotic drivers could have been reinforced mutually promoting restoration (King and Hobbs, 2006), however, longer term studies are needed to assess if grass recovery persists once the rainy period ceases, thus corroborating a state transition (Bestelmeyer et al., 2013).

According to Adámoli et al. (1990), Chaco shrublands may constitute a new steady-state different from the original (dry forest) arising from overgrazing, partial tree depletion and fire suppression. The

starting point towards a shrub dominated steady-state would be the herbaceous layer discontinuity (by overgrazing) plus livestock scarification and dispersion of woody fruits. Reducing grass biomass reduces natural fires that control bush recruitment and survival. Water scarcity is worse in shrublands, where litter cover is lower and insolation and evaporation are higher than in forests (Breshears et al., 1998; Abril and Bucher, 1999), exacerbating the effect of trampling on reducing infiltration. As soil moisture in topsoil horizons is reduced, an asymmetric competition favorable to bushes is established (Cavagnaro and Passera, 1991). Shrub dominance, associated with self-reinforcing

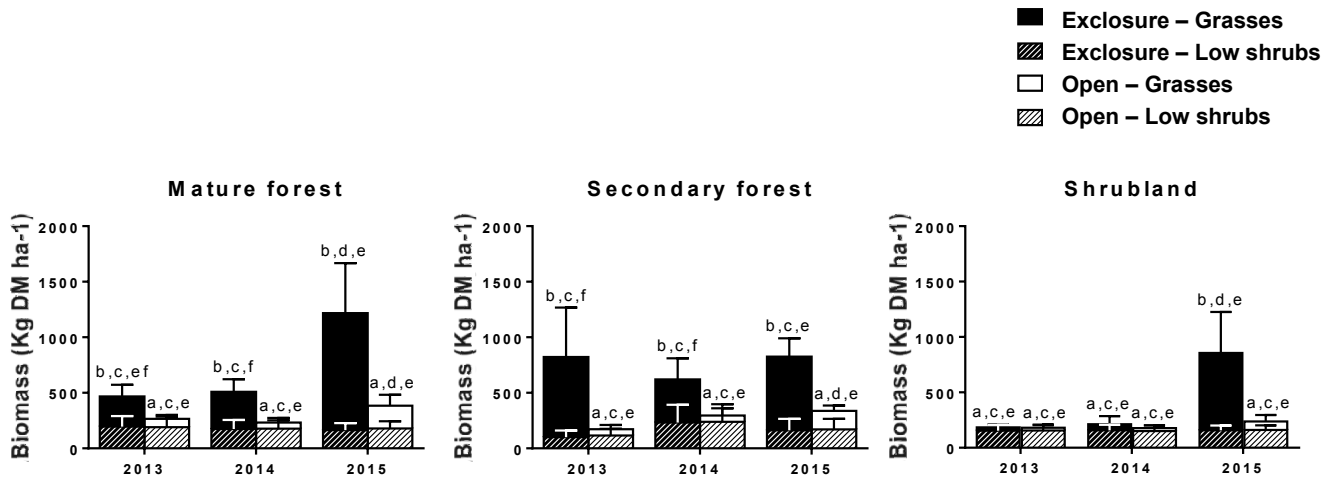


Fig. 2. Peak standing biomass of grasses (flat) and low shrubs (lined) in continuously grazed open areas (white) and seasonal exclusions (black), from the 2nd to the 4th year since the establishment of the exclusions in three patch types of woodland in semiarid Chaco. Bars represent standard deviation. Different letters indicate significant differences ( $P < 0.05$ ) between grazing treatments (first letter: a - b), years (second letter: c - d) within each grazing treatment, and woodlands (third letter: e - f) within each grazing treatment and year.

feedback mechanisms, can hamper grass recovery (various authors in: Laycock, 1991; Ratajczak et al., 2014). Thus, shrublands would have crossed some degradation threshold, related to both biotic interactions and abiotic limitations. In such case, restoration would require not only improved grazing management but also vegetation manipulation (e.g. selective shrub removal, using tree cover to modify micro-environmental conditions) or modification of the physical environment (e.g. adding erosion barriers, manipulating soil cover) (Whisenant, 1999).

In the forests, the herbaceous cover responded positively to grazing rest. However, such response and also the total plant cover were almost equal (even, slightly higher) at intermediate overstory degradation (secondary forest), compared to the lowest degradation level (mature forest). This fact could be explained by the “nutrient-light ratio hypothesis” (Tilman, 1988), which posits that along a gradient of decreasing biomass (or increasing degradation of the overstory), light availability increases but soil resources availability decreases, thus, the overall limitation in the understory would be lower at some intermediate level of the gradient than at any of the ends.

Hence, the ability of exclusions in restoring understory vegetation would be well explained by combining the two models discussed. According to Whisenant (1999), recovery at little stages of low degradation would only require grazing management, since the ecosystem processes are fully functional. Under these conditions, Tilman (1988) hypothesis seems to explain why the highest understory responses do not occur at the extreme of the gradient corresponding to less altered

vegetation states. As degradation increases, primary processes can be damaged or even become nonfunctional, overcoming certain thresholds (Whisenant, 1999). From this point, further removal of trees, instead of favoring grasses by releasing light, would be detrimental to them by exacerbating shrub competition and water shortage.

After four years with grazing exclusion during wet season, rangeland condition was nearly good in both mature and secondary forests (compared to reference values from similar landscape positions and vegetation types in semiarid Chaco; Adámoli et al., 1990; Kunst et al., 2006). The rapid recovery we observed inside these exclusions largely matches the results of previous studies in other semiarid regions (Rosenstock, 1996; Mwilawa et al., 2008; Angassa and Oba, 2010; Verdoodt et al., 2010). Grass establishment occurred in tree covered areas, regardless of the forest type. Under the canopy of trees, litter cover is higher and air and soil temperature are frequently lower, reducing evaporative losses and increasing soil moisture in surface horizons, where grass root systems are located (Joffre and Rambal, 1988). Once established, grasses that remain ungrazed until flowering enrich the seedbed (Tessema et al., 2012), while the increased coverage (living and dead) increases infiltration and creates a more favorable micro-environment for seedlings, self-reinforcing grass recovery (Noy-Meir, 1973).

Only in an extremely wet year (2015), grass biomass increased also in shrubland exclusions. This period coincided with an El Niño Southern Oscillation (ENSO) event, of particular high intensity, which

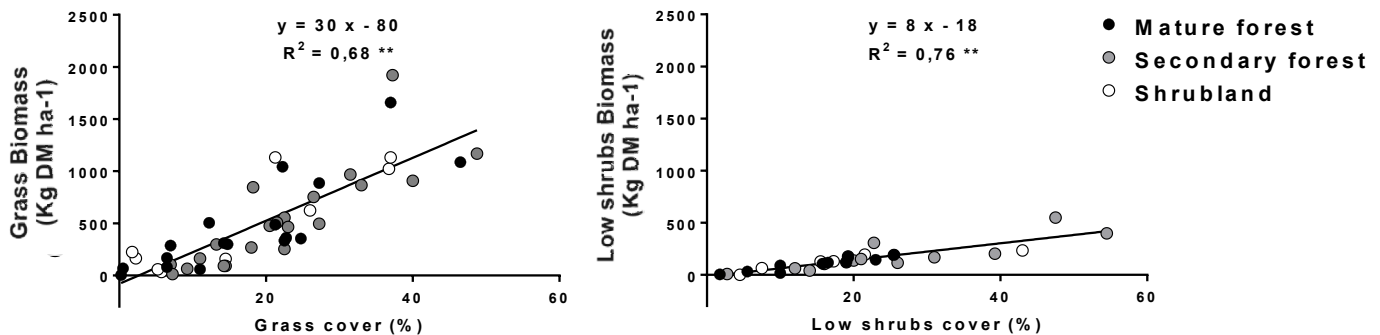


Fig. 3. Relationship between plant aerial cover and peak standing biomass of native grasses (left) and low shrubs (right) in heterogeneous woodlands of semiarid Chaco. Data were gathered in young seasonal exclusions -aged two to four years- from diverse types of patch (mature forests and shrublands: black, grey and white circles respectively) in a varied range of sites and years. The overall correlation between both parameters is shown (x: cover; y: biomass) and its corresponding fit ( $R^2$ ). Asterisks indicate the significance of the regression model (\*\*:  $P < 0.05$ ).

dramatically increases rainfall in certain areas of the world as the Dry Chaco region. ENSO events in combination with grazing control (or low rates of biomass removal) would represent a window of opportunity to start recovery at low biomass states, as shrublands (Holmgren and Scheffer, 2001). Additionally, this wet year (2015) was the only one in which shrub cover decreased in all woodlands, probably because competition became favorable to the grasses. Normal years, with infrequent and intense rains of which a big share infiltrates, are advantageous for shrubs, since they have exclusive access to water deep underground. Conversely, rainy years with more frequent and light rains, are advantageous for grasses, which use water rapidly before infiltrating at deeper soil horizons thanks to its shallow roots and fast growth pulses (Noy-Meir, 1973).

Low shrub cover did not increase in enclosures compared to open areas, contrary to the findings by Yaynesht et al. (2009) and Angassa and Oba (2010), who reported that enclosures increased density and coverage of shrubs and shrub seedlings, especially those most palatable. We suppose that if existed any increase in shrubs could be counterbalanced by winter grazing in seasonal enclosures (Augustine and Mcnaughton, 2004). We assume that, because shrubs (mainly represented in our study by two highly palatable species: *Castela coccinea* and *Celtis tala*) are a major forage resource during dry season (various authors reviewed in: Cotroneo et al., 2016). In accordance, close to our study sites, it was reported that goats grazing at high stocking densities significantly reduced shrub cover (Rueda et al., 2012).

Coinciding with previous works (Abril et al., 1993), we observed a reduction trend in litter accumulation with the increasing overstory degradation, which could lead to reduced soil fertility and moisture retention, and lower seedling establishment (Abril and Bucher, 1999; Verdoort et al., 2009; Raiesi and Riahi, 2014). We did not observe differences in litter cover between grazing treatments, contrary to the findings by Abril and Bucher (1999), who observed significant increases after 8 years of permanent grazing exclusion. Discrepancies could arise from the biomass removal by domestic livestock during the dry season.

Facing the lack of easy methods to estimate forage availability in semiarid Chaco, we obtained biomass–cover regressions for grasses (mostly consumed by cows) and low shrubs (mostly consumed by goats) which, if validated, could offer herders and ecologists a user-friendly tool, inexpensive, rapid and non-destructive, based on visual estimation of aerial coverage. Similar regressions have been previously developed in other dry region by Flombaum and Sala (2007). The linear relationships and the degree of adjustment we found ( $R^2 = 0.68–0.76$ ) match the results reported by them. However, our functional group calibration showed steeper slopes for grasses than for shrubs. This is probably because in the semiarid Chaco, the rains fall during the warm season, while in the system Patagonian arid steppe they studied, rains occur during the cold season. As a consequence, plant growth depends more on actual than on past precipitation in the semiarid Chaco but more on past than on actual precipitation in Patagonia, which would favor Patagonian shrubs. Our results provide relevant information for mixed livestock systems, common in communal dry rangelands. Regressions obtained are useful for different years (wet and dry) and forest types (mature and secondary forests and shrublands) in line with the temporal and spatial variability that characterizes dry systems. However, it must be taken into account that regressions were obtained from (and thus are usable for) relatively low coverages in both cases (0–40%). Linear relationships are typical at low plant densities (Pyke and Archer, 1991), while higher densities and coverage would require different estimation methods.

## 5. Conclusions

This work provides field evidence showing that Dry Chaco shrublands may constitute an alternative steady state from the original forest. This severely degraded state could not be restored by rest from grazing, even when this rest occurs during the growth season. However,

shrublands show signs of recovery when the seasonal grazing rest occurs during the growth season of an extremely wet year. This suggests that water shortage is the main constraint in restoring shrublands. The cover-based estimations of forage availability plus its proper harvesting by managing, could help to maintain an adequate grass litter cover in seasonally grazed areas, improving soil moisture conditions when wet periods ceased and increasing the chances for shrublands to be restored.

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