Impact of 25 years of grazing exclusion and shrub removal on plant community structure and soil characteristics in a xerophytic rangeland

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Abstract

Aim of study: We tested the hypothesis that long periods of grazing exclusion in areas with a history of high grazing intensity will have a positive impact on soil nutrient conditions and favor soil infiltration, increase biomass and lead to a recovery in vegetation.

Area of study: Noria de Guadalupe, Zacatecas, Mexico.

Material and methods: We analyzed the impact of grazing exclusion on biomass, species richness, evenness, soil nutrient content and soil water infiltration after 25 years of exclusion during each of the four seasons by excluding two 15 × 15 m plots of grazing and compared with two control plots.

Main results: Exclusion management did not lead to biomass increases; however, it did lead to an important recovery in the plant community. Moreover, soil nutrient content was more affected by the seasonality of rainfall in the study site than by 25 years of exclusion. The elimination of dominant shrubs in the excluded area led to a faster recovery in palatable shrubs and shortgrass vegetation, which was improved by better infiltration values during the end of spring and summer explaining some of the differences in nutrient availability.

Research highlights: In our study, exclusion management can lead to an important recovery in vegetation without affecting the growth of Atriplex canescens, a valuable source of fodder. Although biomass presented a higher dependence on seasonality and was not related to the treatment, the impact on the forage quality is evident by the different plant communities established after 25 years of exclusion.

Additional key words: Atriplex canescens; community; diversity; grazing; infiltration.

Introduction

An ecological force such as grazing with important environmental and social implications must be analyzed thoroughly to propose sustainable practices (Arévalo, 2012). Grazing management is one of the most important traditional and sustainable land uses in many areas of the world (Milchunas et al., 1988; Crawley, 1997) and grazing lands require appropriate techniques to maintain species composition, soil conservation and high diversity values of plant communities (Baldock et al., 1994; Olff & Ritchie, 1998; Teague & Barnes, 2017). Conversely, mismanagement can cause marked and significant variation in species composition (Casado
et al., 2004; Arévalo et al., 2007). Moreover, overgrazing is a common occurrence in many rangelands, increasing the risks of erosion and desertification and promoting exotic species (Steffens et al., 2008; Gusha & Mugabe, 2013; Encina-Domínguez et al., 2014; Arévalo et al., 2017).

Different grazing studies have often revealed contradictory results on the impact on soil nutrients and species composition (Off & Ritchie, 1998) with conclusions varying from herbivores enhancing to having little effect or negative impacts on plant diversity and richness (Osem et al., 2004; de Bello et al., 2007). Likewise, there are different effects on soil nutrient content (Bakker et al., 2004; Fernández-Lugo et al., 2009; Wang et al., 2016). This lack of consistent responses among studies has been attributed to factors such as the evolutionary history of grazing, productivity gradients, or grazing intensity (Milchunas & Lauenroth, 1993). Thus, to understand the grazing effects on a particular environment, specific studies are required (Perevolotsky & Seligman, 1998). Some authors have related this variability to the stochastic character of grazing ecosystems rather than to deterministic processes (Curtin, 2002). Clearly, measures to avoid degradation of areas due to grazing and adverse environmental conditions are required in many regions such as Africa, where there is strong economic dependence on such grazing activities (Perkins & Thomas, 1993; Thomas et al., 2000). Due to this, in South Africa land management has been directly related to the improvement of grasses and forage legume reinforcement (Tavirimirwa et al., 2012, 2019), grazing management (Gadzirayi et al., 2007), control of grazing pressure (United Nations, 2016) and land tenure reformation systems (Lahiff, 2000).

In many areas of northern Mexico, it is common to have high grazing pressure from goats and cows on grasslands and xerophytic scrub as described by Gutiérrez et al., 2004 or Estrada-Castillón et al., 2010. These authors recommend establishing recovery periods for the biomass. In some cases, some plants highly appreciated by cattle, goats or horses can be promoted, while other shrub species can be removed (Gutiérrez et al., 2007). Other areas with high dependence on grazing pressure provide similar results to those in Mexico and South Africa. In short, overgrazing results in a steady decline in range condition as evidenced by a reduction in forage plants’ palatable quality and in plant species composition (Gusha & Mugabe, 2013). The final situation is a relatively infertile soil and a reduction in the overall productivity of the lands (Toulmin & Scoones, 2001).

In this study, we analyzed the impact of 25 years grazing exclusion in which, as part of the exclusion, all shrubs except Atriplex canescens (Pursh) Nutt. were removed, severing them below the ground level and cutting the roots as much as possible. Atriplex canescens is utilized by farmers in dry ecosystems as maintenance feed for livestock during the drought feed gap or for landscaping purposes (Ventura et al., 2015; Mellado et al., 2018).

The main goal of this exclusion management is to improve soil quality as well as improve biomass production. Thus, we tested the hypothesis that long periods of exclusion (25 years) in areas with high grazing intensity in the past have a positive impact on soil nutrient conditions, favor soil infiltration and increase biomass, regardless of the season. We also postulated that exclusion favor the recovery of plant communities, increasing species richness and diversity.

**Material and methods**

**Study site**

The study was conducted at Noria de Guadalupe, Zacatecas, Mexico (24°21′N, 101°22′W; Fig. 1) at an average altitude of 1,800 m. According to García (2003), the climate is subtropical desert, semiarid with hot summers and cool winters (Papadakis, 1980). Average temperature varies between 7.4 °C in winter and 21 °C in summer with an average annual rainfall of 241 mm, primarily during summer and winter. The average annual temperature is 13.9 °C. The soil is yermosol, slightly alkaline (pH=7.6), poor in organic matter (2.2 ± 0.4%; mean and SD) and nitrogen (53 ± 9.1 kg/ha) and with slight salinity (1.2-2.7 dS/m). Soil depths at the study site range from 66 to 125 cm. Over the study area, the mean percentage of sand, silt and clay is 44.1, 34.9 and 21.0, respectively.

The vegetation type is Chihuahuan Desert Scrub (Henrickson & Johnston, 1986), which is dominated by creosote bush (Larrea tridentata (DC.) Coville) along with tarbush (Flourensia cernua DC.). Other species include mariola (Parthenium incanum Kunth), fourwing saltbush (Atriplex canescens) and scattered colonies of rosetophyllous species such as rough agave (Agave asperrima Jacobi) and lechuguilla (Agave lechuguilla Torr.). The main perennial grasses are burro grass (Scleropogon brevifolius Phil.), alkali sacaton (Sporobolus airoides (Torr.) Torr.) and fluffgrass (Munroa pulchella (Kunth) L.D. Amarilla). The most abundant forbs are desert zinnia (Zinnia acerosa (DC.) A. Gray), Fendler’s bladderpod (Physaria fendleri (A. Gray) O’Kane & Al-Shehbaz), fiveneedle pricklyleaf (Thymophylla pentachaeta (DC.) Small), spear globe mallow (Sphaeralcea hastulata A. Gray) and dwarf...
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The last 25 years. Consequently, the area is dominated by a scrub vegetation community with low diversity and low aboveground biomass (Mellado et al., 2012). During the winter of 1987-1988, an experimental degraded site was chosen in which all shrubs except those of *Atriplex canescens* were removed to favor colonization of the native plant community, while maintaining this appreciated shrub for grazing animals. The other shrubs were removed by severing them below ground level. After that, roots were cut as much as possible with a pickaxe and manually removed from the 25-ha (the experimental site). The root balls of shrubs were pulled out using a spade, shovel and a mattock.

Figure 1. Location of the study site (area enclosed by the black line), Noria de Guadalupe, situated at the northeast of the state of Zacatecas, Mexico.

In the last five decades, the area has been managed to support communal grazing of horses, cattle, sheep and goats with a stocking rate above the grazing capacity recommended for this vegetation type in its present state of plant community degradation (loss of species and biomass). The stocking rate in the area per animal unit (AU) has been estimated in 30-40 ha/AU (assuming that each AU is a representative cow of 450 kg or its standardization to other animals such as horses or goats) (Gutiérrez et al., 2007), based on the local information provided by rangers on this area, which has remained relatively constant along the last 25 years. Consequently, the area is dominated by a scrub vegetation community with low diversity and low aboveground biomass (Mellado et al., 2012). During the winter of 1987-1988, an experimental degraded site was chosen in which all shrubs except those of *Atriplex canescens* were removed to favor colonization of the native plant community, while maintaining this appreciated shrub for grazing animals. The other shrubs were removed by severing them below ground level. After that, roots were cut as much as possible with a pickaxe and manually removed from the 25-ha (the experimental site). The root balls of shrubs were pulled out using a spade, shovel and a mattock.
Design of the study

In 1987, in the 25-ha study site (where all the shrubs but *Atriplex canescens* were removed), two 20 × 20 m permanent exclosures were established to prevent grazing. Inside each exclosure, a 15 × 15 m plot was established, located in the center. Two control plots (outside of the exclosures) of the same size (15 × 15 m) were randomly established in the experimental site (on the 25-ha) as controls with free grazing of goats, cattle and horses. These plots were situated more than 100 m far from the fence and from any trails or roads, avoiding areas with evident human disturbance. Both plots were representative of the extension of the study area and treatments.

Plant and soil traits were measured four times on the year 2012 at the end of each season (end of June, September, December and March) in order to provide data on periodical changes of vegetation composition, biomass, soil composition and infiltration. Seasons are remarkably different in this area (especially humidity), so the seasonal effect on those parameters should be analyzed to reveal how important they are with respect the exclusion. As we were interested in the change of biomass, species composition, soil nutrient composition and infiltration after 25 years of exclusion, the control provided the baseline of these changes.

For each plot and season, we collected ten soil samples of 1-kg at 0-30 cm depth (avoiding the biomass subplots). Each sample was mixed, dried and sieved through a 2-mm sieve, and debris and stones were removed from the soil. We analyzed pH, Olsen P (mg/L), total N percentage and NO₃⁻ (mg/L), percentage of or -

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Results

Total biomass in the different plots varied between seasons (Pseudo F3,56 = 78.03, p<0.01) but not for the exclosure treatment (Pseudo F1,56 = 2.89, p = n.s.) or interaction among factors (Pseudo F3,56 = 0.94, p = n.s.). The highest value of biomass was found in summer (Fig. 2a). As regard richness, a total of 22 plant species were recorded, 20 in the exclosure plots and only 9 in the grazed plots (Table 1 and Table S1 [suppl]).

Two species were only found in the grazed plots: *Koeberlinia spinosa* Zucc. and *Cylindropuntia leptocaulis* (DC.) F.M. Knuth. Other species were present in both treatments. *Atriplex canescens* presented high biomass after 25 years in both areas, grazed and excluded, maintaining high values.

Richness varied significantly among seasons (Pseudo F3,56 = 24.11, p<0.01) and between exclosure and grazed plots (Pseudo F1,56 = 75.85, p<0.01) as did the interaction between seasons and exclosure (Pseudo F3,56 = 24.11, p<0.01; Fig. 2b). For evenness, differences were only found for seasons (F3,56 = 3.14, p<0.05) but not for the exclosure treatment (F1,56 = 1.75, p = n.s.) or interaction (Pseudo F3,56 = 0.41, p = n.s.).

The average soil nutrients analyzed are presented in Table 2. We were interested in revealing differences between treatments and seasons on nutrient composition. PCA of nutrients did not discriminate between grazed and exclosure plots. In spite of this, Fig. 3a, in the axis I, revealed a strong relationship with NO3 content, while axis II is more related to Ca and Mg and Zn and Cu. However, it is possible to discriminate the plots based on season and exclosure in the bidimensional space of axes I and II. Spring sample plots have a higher content of Ca and Mg, and lower values of Zn and Cu. For the rest of the seasons, the discrimination is very poor, revealing no clear patterns (Fig. 3b).

DCA revealed strong differences in species composition between grazed and exclosure plots (Fig. 4a). As well as evidence of higher richness in exclosure plots, it was also found that shrubs such as *L. tridentata*, *C. leptocaulis*, *Prosopis glandulosa* Torr. and *K. spinosa* are the dominant species in the grazed vegetation. In the case of grasses in exclosure plots, some grasses, such as *Bouteloua gracilis*, *Muhlenbergia repens*, *Aristida adscensionis*, *Achnatherum editorum* or *Eragrostis* sp., are dominant in the community. This pattern is maintained for the different seasons, with poor discrimination among the highly dominant species between grazing vs. exclosure treatments (Fig. 4b).

The results of infiltration (Fig. 5) indicate significant differences for summer (Z=-2.52, n=8 and p<0.05) with higher values of infiltration in the exclosure plots, whereas for winter and spring there were higher values of infiltration in the grazed plots (Z = -2.38 and Z = -2.30, n=8 and p<0.05 respectively). With respect to autumn, no significant differences were found.

Discussion

The vegetation of the studied area can be considered overgrazed (Gutiérrez *et al.*, 2007) with an increase in the desertification level (dominant desert and non-palatable species becoming more common), but not a significant reduction in biomass. Instead, significant differences in seasonal biomass production were recorded, especially in summer (Fig. 2a).
After 25 years of livestock exclusion, we expected an increase on aboveground biomass, as has been confirmed in other studies (Castro & Freitas, 2009; Wang et al., 2016). However, this was not the case in this study probably because overgrazing aboveground biomass of grasses and forbs decrease as shrubs increase.

### Table 1. Average species biomass (N=4) in each season and for each treatment (kg/ha dry biomass).

| Species            | Exclusion | Grazed |
|--------------------|-----------|--------|
|                    | Spring    |        |
| Atriplex canescens | 97.5      | 65.3   |
| Bouteloua gracilis| 12.5      | -      |
| Eragrostis sp.     | 20.0      | -      |
| Larrea tridentata  | 15.0      | 163.5  |
| Prosopis glandulosa| -        | 37.5   |
| Scleropogon brevifolius| 10.0 | -      |
| Tecoma stans       | 20.0      | -      |
| Other species      | 5.0       | -      |
|                    | Summer    |        |
| Achnatherum editorum| 33.8 | -      |
| Ambrosia confertiflora| 30.0 | -      |
| Atriplex canescens | 355.0     | 273.8  |
| Bouteloua gracilis| 31.3      | -      |
| Buddleja scordioides| 47.1 | -      |
| Eragrostis sp.     | 35.6      | -      |
| Gutierrezia sarothrae| 50.0| -      |
| Koeberlinia spinosa| -       | 50.0   |
| Larrea tridentata  | 52.5      | 408.1  |
| Lepidium virginicum| 22.5  | -      |
| Marrubium vulgare  | 32.5      | -      |
| Muhlenbergia repens| 23.5  | -      |
| Munroa pulchella   | 16.3      | -      |
| Parthenium incanum | 27.5   | 15.0   |
| Physaria fendleri  | 17.1      | 13.3   |
| Scleropogon brevifolius| 35.4 | -      |
| Sporobolus airoides| 106.7  | -      |
| Tecoma stans       | 11.4      | -      |
| Other species      | 5.0       | -      |
|                    | Autumn    |        |
| Achnatherum editorum| 20.0  | -      |
| Ambrosia confertiflora| 15.0  | -      |
| Aristida adscensionis| 10.0 | -      |
| Atriplex canescens | 262.5     | 131.9  |
| Bouteloua gracilis| 25.0      | -      |
| Buddleja scordioides| 6.7   | -      |
| Eragrostis sp.     | 17.5      | -      |
| Koeberlinia spinosa| -       | 40.0   |
| Larrea tridentata  | 28.8      | 260.0  |
| Physaria fendleri  | -         | 6.7    |
| Prosopis glandulosa| 167.5  | -      |
| Scleropogon brevifolius| 10.0 | -      |
| Sporobolus airoides| 30.5   | -      |
| Tecoma stans       | -         | 17.5   |
|                    | Winter    |        |
| Atriplex canescens | 95.0      | 52.5   |
| Bouteloua gracilis| 5.0       | -      |
| Cylindropuntia leptocaulis| - | 25.0 |
| Eragrostis sp.     | 12.5      | 3.3    |
| Koeberlinia spinosa| -       | 25.0   |
| Larrea tridentata  | 7.5       | 143.8  |
| Prosopis glandulosa| 37.5   | 32.5   |
| Scleropogon brevifolius| 5.0  | -      |
| Sporobolus airoides| 10.0   | -      |
| Other species      | 5.0       | -      |

*Figure 3. (a) Principal components analysis (PCA) of the soil samples of the plots and soil nutrients. Arrows indicate variables used. We used the information of five samples obtained from each subplot of both plots of the treatment and at each station. Samples of the same treatment are enclosed in a polygon (excluded plots: solid line; grazed plots: dashed line; eigenvalue for axis I: 0.87; eigenvalue for axis II: 0.11; cumulative percentage of variance explained for axes I and II: 97%). (b) In the same PCA analysis, we discriminated the plots based on season (spring, summer, autumn and winter) and treatment (excluded vs. grazed plots). Black polygons are used for grazed enclosing subplots at different seasons, while grey color is used for excluded polygons; summer: thin solid line, autumn: dotted line, spring: dashed line and winter: solid line.*
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Atriplex species are associated with areas of high salinity, low humidity and high temperature (Ramos et al., 2004). They have been used for erosion control and rehabilitation in salt-affected and degraded areas (de Souza et al., 2012; Rani et al., 2013). In desert regions, they have been used by farmers as maintenance feed for livestock during the drought feed gap (Norman et al., 2010; Pearce et al., 2010) or for landscaping purposes (Panta et al., 2014; Ventura et al., 2015).

Soil nutrient content composition changes related to seasonal variability and grazing vs. excluded treatment do not show any discrimination among plots (Figs. 3a,b). Although in many areas the impact of grazing shows important differences (Arévalo, 2012), we found that the most relevant factor in soil nutrient composition is related to seasonality. Based on the analysis (Fig. 3b), Ca and Mg showed higher values at the end of spring. In some areas, the increase in Ca and Mg is related to the solubility of these available cations in water (Chapman et al., 1997; Walna et al., 1998) or is even due to incorporation through rain (Sapek, 2014). From June to September, more than 45% of the total precipitation occurs in the study area, with July being the wettest month (77 mm) and March the driest in this period (12.4 mm; SMN, 2010). We can relate this cation solubilization to the first considerable precipitations of the year, as the soil was sampled at the end of June. This is supported by the infiltration values obtained (Fig. 5), compensating for the decrease in grass and forb production (Aguir et al., 1996). Several other studies have also indicated a low impact on aboveground plant biomass after several years of grazing exclusion (Arévalo et al., 2011a). However, it is important to note that the biomass of the unpalatable species is dominant in the grazed areas (Table S1 [suppl]).

As other studies have revealed (Karakosta & Pap-anastasis, 2007; Strand et al., 2014), an increase in woody species in grazed areas has been related to an increase in wildfire risk (Arévalo, 2012; Arévalo & Naranjo, 2018). However, regular patterns are still difficult to establish (Hughes et al., 2006).

In the study area, species richness was particularly vulnerable to grazing, something that we ascribe to overgrazing activity carried out in the area. Indeed, it is a common result in many overgrazed areas (Al-Rowaily et al., 2012; Wang et al., 2016; Zhu et al., 2016). However, these results are difficult to extrapolate, as they largely depend on the characteristics of the plant community and soil conditions, as well as other human disturbances (Olff & Ritchie, 1998). As mentioned, many studies have shown positive results of grazing on diversity (Tälle et al., 2016). In the present study, evenness index did not reveal differences. The highest species richness is found in late spring and summer (humid period), revealing its relationship with precipitation and soil infiltration.

Although A. canescens was not removed in the study site, it is not a representative species in the community in the exclosure plots as it maintains high biomass values throughout the seasons in both areas (grazed vs. excluded). Atriplex species are associated with areas of high salinity, low humidity and high temperature (Ramos et al., 2004). They have been used for erosion control and rehabilitation in salt-affected and degraded areas (de Souza et al., 2012; Rani et al., 2013). In desert regions, they have been used by farmers as maintenance feed for livestock during the drought feed gap (Norman et al., 2010; Pearce et al., 2010) or for landscaping purposes (Panta et al., 2014; Ventura et al., 2015).

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| Table 2. Mean and standard deviation (SD) of the nutrients in the subplots (n=10) in each season and for each treatment (E: excluded, G: grazed). |
|----------------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|----------------|
|               | pH             | %OM       | %TN         | Ca (meq L⁻¹) | Mg (mg/L)   | Na (mg/L)   | P (mg/L)    | NO₃ (mg/L) | Cu (mg/L) | Zn (mg/L) |
| Spring_E      | 7.68 ± 0.05    | 2.31 ± 0.44 | 0.14 ± 0.02 | 35.80 ± 1.90 | 4.08 ± 1.24 | 2.54 ± 0.99 | 6.78 ± 1.03 | 115.08 ± 8.84 | 1.61 ± 0.05 | 1.94 ± 0.07 |
| Spring_G      | 7.67 ± 0.09    | 2.76 ± 0.35 | 0.12 ± 0.04 | 33.12 ± 6.08 | 6.04 ± 0.89 | 3.21 ± 0.80 | 11.00 ± 8.49 | 88.49 ± 10.89 | 1.65 ± 0.14 | 2.10 ± 0.13 |
| Winter_E      | 7.76 ± 0.14    | 2.31 ± 0.55 | 0.13 ± 0.02 | 18.08 ± 1.98 | 2.76 ± 0.57 | 3.19 ± 1.55 | 18.27 ± 24.13 | 81.95 ± 24.13 | 1.89 ± 0.21 | 1.95 ± 0.13 |
| Winter_G      | 7.83 ± 0.11    | 2.39 ± 0.44 | 0.12 ± 0.02 | 15.08 ± 8.81 | 3.64 ± 1.27 | 2.44 ± 0.54 | 22.18 ± 9.81 | 131.53 ± 131.53 | 1.83 ± 0.14 | 1.96 ± 0.17 |
| Summer_E      | 7.71 ± 0.17    | 1.67 ± 0.33 | 0.16 ± 0.01 | 11.59 ± 4.10 | 2.18 ± 0.52 | 2.04 ± 0.62 | 13.95 ± 4.10 | 130.80 ± 4.10 | 6.55 ± 0.10 | 2.45 ± 0.12 |
| Summer_G      | 7.64 ± 0.18    | 2.23 ± 0.31 | 0.11 ± 0.04 | 6.50 ± 1.40  | 1.58 ± 0.34 | 2.00 ± 0.59 | 14.97 ± 3.28 | 132.38 ± 3.28 | 6.40 ± 0.07 | 2.47 ± 0.12 |
| Autumn_E      | 7.53 ± 0.17    | 2.61 ± 0.38 | 0.14 ± 0.04 | 12.09 ± 8.09 | 2.00 ± 0.43 | 2.59 ± 0.11 | 7.04 ± 19.67 | 80.94 ± 19.67 | 6.27 ± 0.05 | 2.44 ± 0.12 |
| Autumn_G      | 7.58 ± 0.39    | 2.69 ± 0.47 | 0.13 ± 0.02 | 6.39 ± 5.12  | 1.80 ± 0.75 | 2.61 ± 0.19 | 6.01 ± 7.85  | 78.53 ± 7.85  | 6.29 ± 0.02 | 2.39 ± 0.40 |

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revealing significant differences in summer between the two areas once precipitation has become more abundant. Many factors are involved in water infiltration as saturated conductivity and infiltration capacity (Mein & Larson, 1973) as well as rain intensity. These factors determined that all the rain infiltrated into the factors due to the desert characteristics of the area. The excluded area showed...
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The study of grazing exclusion has potential management implications (e.g. setting the maximum number of animals, restricting traditional activities in the protected area, etc.) and environmental implications (e.g. plant and animal protection, control of invasive species…). It also has great socioeconomic importance because such research provides useful information to help range managers to continue grazing activity, which is of high cultural value (Arévalo et al., 2011a, b).

In our study, despite the design of the experiment being somewhat limited, it was based on a long-term period of exclusion and a much extended area of plots. Therefore, we consider that the results are consistent and provide information that can be of interest. Exclusion management can lead to an important recovery in vegetation without affecting the growth of *Atriplex canescens*, a valuable source of fodder. Although biomass presented a higher dependence on seasonality and was not related to the treatment, the impact on the forage quality is evident by the different plant communities established after 25 years of exclusion. Therefore, due to the recovery of the plant community in the excluded area, we suggest that a reduction in grazing intensity is necessary to regain the productivity of palatable plants. Reducing grazing intensity also favors soil infiltration in summer, which, in turn, favors productivity and increases species diversity. The removal of shrubs can accelerate recovery, but leaving *A. canescens* untouched seems to have had no effect on its dominance in the area. Thus, in order to protect and favor these traditional activities, we suggest controlling

higher infiltration in summer, but significantly lower in winter and spring. As long as rain appears at the end of spring and summer in this area, we can assume that it will have an impact on the solubilization as availability of cations, as well as on the water available for the vegetation.

Species composition revealed important changes in the plant community determined by exclusion, whereas there was low variability with respect to the species composition determined by season (Figs. 4a,b). Scrub species dominated the grazed vegetation, while the excluded area was dominated by grasses and forbs, which is a common result (Karakosta & Papanastasis, 2007; Mayer et al., 2009). Grazing on more palatable species can sometimes promote dispersion, but in overgrazed areas the impact is almost a complete elimination of these species (Gusha & Mugabe, 2013), leaving only a plant community dominated by shrubs of low palatability and with an array of mechanical defenses against herbivores. This was the case in this study with three non-palatable species like *L. tridentata*, *C. leptocaulis* and *K. spinosa* dominating the exclusion area, while the excluded area was dominated by palatable grasses species as *B. gracilis*, *M. repens*, *A. editorum* or *Eragrostis* sp. as well as shrubs such as *Gutierrezia sarothrae* and *Buddleja scordioides*. Thus, it appears that exclusion leads to a partial recovery of the dominant plant community of the short grass. Reducing grazing intensity would likely favor a slight recovery of degraded grazing land, which is an important economic resource for farmers in the area (Teague & Barnes, 2017).

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grazing on this vegetation through management of grazing pressure, as a way of preserving landscape uses, cultural values and biodiversity.

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