

Vegetation change in response to grazing exclusion in montane grasslands, Argentina

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Background and aims – Natural montane grasslands in the Southern Pampas of Argentina are expected to show a high potential for recovery after heavy grazing due to their evolutionary history in the presence of large herbivores and their high productivity. The objective of this work is to compare plant diversity, bare soil percentage, biomass and botanical composition between grazed and non-grazed areas at different times following grazing exclusion.

Methods – Vegetation was assessed on exclosures established in 2006 and on nearby areas open to grazing by feral horses in December 2006, 2007, 2008 and 2009. In 2009 we added 15 year old exclosures to the analysis.

Key results – Species richness declined 24 months after grazing exclusion, mainly due to a decrease in the abundance of forb species. Bare soil percentage was significantly reduced 12 months after exclosures were built. Above-ground biomass showed a 3-fold increase 12 months after grazing exclusion. After three years, species richness and biomass were similar to those corresponding to older exclosures (15-years old). The composition of plant communities also changed following horse exclusion, with three and 15-years old exclosures dominated by perennial grasses typical of late seral stages.

Conclusions – Our results support the hypothesis that natural montane grasslands in the Southern Pampas of Argentina may recover fast from grazing by large herbivores without application of specific restoration techniques.

Key words – bare soil, drought, feral horses, grassland diversity, grassland recovery, natural grassland, overgrazing, vegetation changes.

INTRODUCTION

Herbivores play an important role in molding vegetation structure and biodiversity in grasslands worldwide (McNaughton 1984, Olff & Ritchie 1998). The response of grasslands to the presence of herbivores depends on their evolutionary history of grazing and their productivity (Milchunas et al. 1988). Ecosystems that have evolved with presence of large herbivores are expected to exhibit a high recovery potential in species richness after intensive grazing (Cingolani et al. 2005). In ecosystems that have evolved with low grazing pressures, on the contrary, the presence of large herbivores could lead to transitions among different meta-stable states (Westoby et al. 1989, Carpenter et al. 2001, Briske

et al. 2008), eventually driving to irreversible community changes (Cingolani et al. 2005).

Vegetation response to grazing has been extensively studied in natural and semi-natural grasslands (Gillen et al. 1998, Chaneton et al. 2002, Osem et al. 2002, Hayes & Holl 2003, Loucougaray et al. 2004, Rook et al. 2004, Cesa & Paruelo 2011, Zhao et al. 2011), however, little is known about short-term response of vegetation to grazing exclosure in natural grasslands. This kind of information gives valuable insights that could help in deciding the eventual need of complementary restoration actions in protected areas (Cuevas & Zalba 2010) as well as management advice for cattle production.

Grazing by large herbivores is generally associated with small prostrate plants, annuals species and unpalatable grasses (i.e. undesirable) (Lavorel et al. 1999, Díaz et al. 2001, Peco et al. 2005, Díaz et al. 2007). Grazing abandonment may lead toward different community composition and particularly overgrazing may lead to the creation of alternative undesirable states (Westoby et al. 1989). This shift toward alternative states depends on grazing regime (e.g. herbivore type, intensity) and environmental conditions (e.g. precipitations, fire regime) (Cingolani et al. 2005). After abandonment, grasslands are rapidly dominated by grasses and tall species with a consequent loss of biodiversity (Galvánek & Lepš 2008, Liira et al. in press). Generally, long-term abandonment leads to forest communities in semi-natural grasslands (Hansson & Fogelfors 2000). But in natural grasslands, as Pampa grasslands, absent of herbivores may lead to less productive grasslands subject to a high fire frequency and dominated by unpalatable species (de Villalobos & Zalba 2010, Tizón et al. 2010, Peter et al. in press), or facilitates the invasion of non-native woody species (Chaneton et al. 2002, Zalba & Villamil 2002).

Pampa grasslands in Argentina had been changing during at least the last 130 years, with the introduction of agriculture and cattle (Hudson 1929, Bilenca et al. 2009). However, in the last two decades, new crops (e.g. GMO soybean) and new farming techniques (e.g. no-till farming) are displacing cattle raising to relictual natural grassland areas, which has led to overgrazing in these sites (Oosterheld et al. 1992, Latterra et al. 2009). Grassland biodiversity is endangered since c. 1.05% of these areas are protected (Bilenca & Miñarro 2004). Additionally, ecosystems services provided by grasslands are being affected. For example, soil erosion due to the lack of a vegetal cover may reduce productivity, water infiltration and fresh water supply in these areas (Latterra et al. 2009, Chartier et al. 2011). However, there are still few remnants of more pristine communities on the mountain situated in the southern part of the Pampas. Here above-ground annual primary production reaches c. 500 g.m⁻² (Pérez & Frangi 2000), although this may vary depending on rainfall (Frangi et al. 1980, Pérez & Frangi 2000). The Pampas evolved subjected to grazing by native herbivores (Hudson 1929, Lauenroth 1998), being the guanaco (*Lama guanicoe*) the most abundant species during the past several thousand years (Lauenroth 1998). Unfortunately, there is no documentation of population levels or grazing intensity of guanacos. Recurrent fires and droughts are other structuring forces in this ecosystem (Barrera & Frangi 1997), being both cited as promoters of exaptation to grazing (Coughenour 1985). Since European settlement, and especially during the last century, native herbivores were replaced by domestic and feral herbivores (i.e. horses, cattle and sheep), and natural disturbances, such as fire, were highly reduced or even suppressed (Brailovsky & Fogelman 1998, Bilenca & Miñarro 2004). Medina (2007) reported a fire-free interval of c. 4 years in nearby areas (< 300 km) for the last 200 years.

Based on these conditions and on current theory, Ventania montane grasslands are expect to rapidly recover from grazing by large herbivores. However there is no information available that indicates if it is necessary to apply restoration techniques in this type of grasslands. In order to test this, we

compared plant species richness, bare soil percentage, biomass and botanical composition between areas grazed by feral horses and exclosures of different age.

METHODS

Study area

The study area was located at Ernesto Tornquist Provincial Park (ETPP), east-central Argentina (38°03'S 61°58.5'W). Climate is sub-humid temperate (Burgos & Vidal 1951). Average annual air temperature is 14°C and annual precipitation 800 mm (Burgos 1968). Annual precipitation in ETPP during the study period was 858 mm, 954 mm, 690 mm and 621 mm, in 2006, 2007, 2008 and 2009, respectively. During the study period the area was subjected to moderate drought conditions from autumn 2008 to autumn 2009. Natural fires and drought periods are common in the area (Barrera & Frangi 1997).

Soils in the study area are classified as Lytic Hapludolls and Argidolls, and characterized by high organic content (c. 7%) on superficial horizons (Cappannini et al. 1971, Frangi et al. 1980). In the absence of grazing by non-native herbivores, the physiognomy of the vegetation is grassland with sparse small shrubs and without bare soil patches. Grass canopy reaches 50 to 60 cm in height and is dominated by *Piptochaetium hackelii*, *Nassella melanosperma* and *Chascolytrum subaristatum* (Frangi & Bottino 1995).

ETPP has a recent history (around thirty years) of heavy, continuous, year-round grazing by feral horses. Feral horses reached a maximum density of 32.5 horses per km² in early 2002 (Scorolli & Cazorla 2010). After that, the population was reduced by mass mortality caused by extreme windy and rainy conditions during a lightning storm in November 2002 (Scorolli et al. 2006), and due to planned retirement of horses in 2006 (Smorzeňuk 2008). In 2008 the feral horse population was reduced to a minimum density of c. 7.25 horses per km² (Smorzeňuk 2008) and slowly recovered thereafter (A. Scorolli, Universidad Nacional del Sur, Argentina, pers. comm.). Outside the ETPP similar grasslands are grazed by cattle. Before the introduction of non-native large herbivores, the study area was grazed by native ungulates, particularly guanaco (*Lama guanicoe* Müller) and Pampas deer (*Ozotoceros bezoarticus* L.) (Hudson 1929, Bilenca & Miñarro 2004).

Experimental design and sampling

In November 2006 we randomly located one 20 × 50 m² area on three different watersheds separated by 250 m or more from each other. These areas had similar orientation (E-NE), slope (c. 5%), altitude (450–500 m a.s.l.), soil conditions and botanical composition, and were located at those areas of the study area that were most commonly used for grazing by horses (Scorolli 2007). Each of these areas was divided in two 20 × 20 m² plots with a 10 × 20 m² buffer area between them, and randomly assigned to one of two treatments: grazing exclusion or grazing. Exclosures were constructed with fences made of barbed wire. Each pair were considered as a block (n= 3), leading to a completely randomized block de-

sign. Samples taken within each plot were grouped to obtain only one value per treatment per block. Vegetation relevés were made immediately after (December 2006) and 12, 24 and 36 months after building the exclosures. This sampling date (i.e. December) was chosen since biomass production peaks at the end of spring (Frangi et al. 1980, Pérez & Frangi 2000). Vegetation cover and bare soil percentage were sampled on five 1 m² sub-plots randomly placed in each plot, using a modification of the Braun-Blanquet dominance scale (Sutherland 1996), with 10% cover intervals for vegetation and 5% cover intervals for bare soil. Above-ground biomass samples were harvested by clipping the vegetation at ground level in three 0.2 m² subplots that were randomly placed inside every plot. Biomass was washed to remove soil particles and oven-dried until reaching a constant weight. Additionally, in December 2009, vegetation relevés and biomass samples were taken from three 20 × 20 m² plots randomly placed inside a 28 ha, 15-years old exclosure in the proximity (< 500 m away) of those established in 2006. These last plots shared the same orientation, slope, altitude and soil conditions with those established in 2006. They were separated from each other at least 250 m apart to reduce pseudo replication problems. This exclosure is the only available site in the study area that had been closed to feral horses for a long period.

Statistical analysis

We analyzed temporal changes in species richness, biomass and bare soil percentage in the different treatments by using mixed models analysis and repeated-measures approach (Littell et al. 2002). The best model for covariance was selected following the Akaike's Information Criterion. For details see Pinheiro & Bates (2000) and Di Rienzo et al. (2010). As interaction between treatment (exclosure/grazing) and time (0/12/24/36 months after grazing exclusion) were significant in all cases, both factors were analyzed separately (Quinn & Keough 2002). Treatment effect was tested with paired t tests for every sampling date, whereas the effect of time was assessed by means of one-way ANOVA with a block design for every treatment. Prior to analysis, species richness was square-root transformed and biomass and bare soil data were natural logarithm transformed to fulfill ANOVA assumptions (Zar 1999). Tukey test ($\alpha = 0.05$) was used for mean comparisons. In December 2009 we followed the same procedure to sample vegetation in the 15-years old exclosure and compared species richness and above-ground biomass between 3-years and 15-years old exclosures using one-way ANOVA. All analyses were made using InfoStat (Balzarini et al. 2008).

We performed a hierarchical cluster analysis in order to search for changes in the botanical composition of these grasslands and we used Indicator Species Analysis to optimize the number of clusters, as described by McCune & Grace (2002). For the cluster analysis, sample cases were vegetation relevés in every exclosure and grazed area in different times of the study period ($n = 27$) and species cover was used as variables ($k = 121$). The cluster analysis was conducted using Bray-Curtis distance measure with unweighted arithmetic average clustering method (UPGMA)

(Legendre & Legendre 1998). All data were arc-sine square root transformed before analysis. The dendrogram was scaled by Wishart's objective function and converted into a scale of percentage of information remaining (i.e. when all groups are fused, there is no information remaining) (McCune & Meford 1999). We performed an Indicator Species Analysis, using the Dufrene and Legendre's method (Dufrene & Legendre 1997), at each step of cluster formation. Indicator values (IV) and their associated probabilities (p-values) were then calculated for each species at each level of grouping, only for those species occurring in three or more plots. We averaged the resulting p-values across all species for each step of clustering. The cluster step yielding the smallest average p-value was taken as the cut-off (pruning) value. This was at c. 55% of information remaining. For details see McCune &

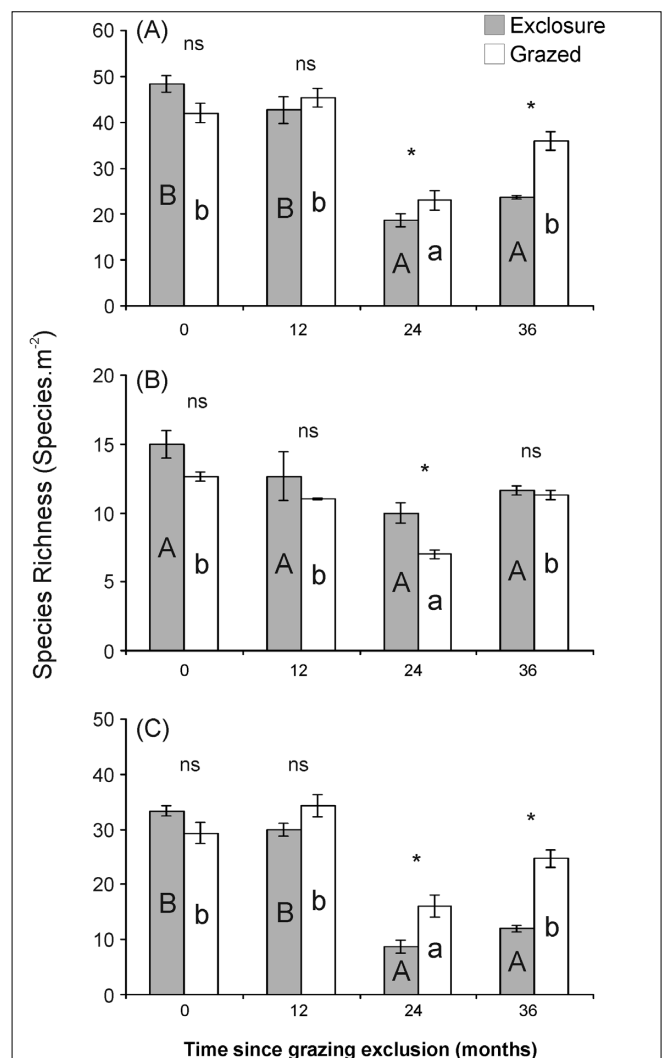


Figure 1 – Temporal changes in total (A), grasses (B) and forbs (C) species richness in exclosures and grazed areas. ns: non-significant differences between treatments, *: significant differences ($p < 0.05$) between treatments. Different letters on bars are significantly different mean values (Tukey test at $p < 0.05$). Capital letters are used for non-grazed areas and lower-case letters for grazed areas. Bars represent ± 1 standard error.

Table 1 – Indicator species of vegetation relevés.

Percentage cover and indicator species value (IV) of established vegetation from all detected groups (I, II III and IV) are shown. Only the most important species are shown (IV>50). Origin (according to Zuloaga & Morrone 2007): E: endemic species, I: non-native species, N: native species. A: annual species, P: perennial species.

Species (Family)	Origin	Life Cycle	Group	IV	Mean Cover per group (%)			
					I	II	III	IV
<i>Anemone decapetala</i> (Ranunculaceae)	N	P	III	70	< 1	0	1	0
<i>Aristida spegazzinii</i> (Poaceae)	N	P	II	55.6	10	26	13	< 1
<i>Briza minor</i> (Poaceae)	I	A	III	87.5	< 1	0	1	0
<i>Eragrostis lugens</i> (Poaceae)	N	P	IV	44.6	< 1	< 1	0	5
<i>Eryngium paniculatum</i> (Apiaceae)	N	P	IV	40.9	5	1	1	9
<i>Geranium albicans</i> (Geraniaceae)	E	P	I	90.2	1	< 1	0	0
<i>Hypochaeris radicata</i> (Asteraceae)	I	P	II	100	0	10	0	0
<i>Lolium multiflorum</i> (Poaceae)	I	A	III	85	< 1	0	1	0
<i>Medicago minima</i> (Fabaceae)	I	A	III	83.3	0	0	1	0
<i>Melica argyrea</i> (Poaceae)	E	P	IV	42.7	2	2	< 1	5
<i>Mimosa rocae</i> (Fabaceae)	E	P	II	84.1	< 1	5	< 1	0
<i>Nassella trichotoma</i> (Poaceae)	E	P	IV	44.2	2	2	< 1	7
<i>Oxalis articulata</i> (Oxalidaceae)	N	P	III	58.7	< 1	< 1	1	0
<i>Pelletiera verna</i> (Primulaceae)	N	A	III	92	< 1	0	6	0
<i>Piptochaetium hackelii</i> (Poaceae)	E	P	IV	59.7	< 1	< 1	8	27
<i>Piptochaetium montevidense</i> (Poaceae)	N	P	III	64.9	3	2	17	< 1
<i>Piptochaetium stipoides</i> (Poaceae)	N	P	I	37.1	33	17	14	9
<i>Plantago myosuros</i> (Plantaginaceae)	N	A	I	84.2	2	0	0	< 1
<i>Scleranthus annuus</i> (Caryophyllaceae)	I	A	II	50.1	1	2	< 1	< 1
<i>Silene gallica</i> (Caryophyllaceae)	I	A	I	68.4	2	< 1	< 1	< 1

Grace (2002). All analyses were made using PC-ORD 5.33 (McCune & Mefford 2006).

Nomenclature

Botanical nomenclature of all taxa follows the criteria used in the Catálogo de Plantas Vasculares del Cono Sur (Zuloaga & Morrone 2007). The names of the families used are those preferred in more recent publications instead of traditional designations.

RESULTS

Changes in species richness, bare soil and above-ground biomass

Total, grass and forbs species richness were not statistically different between treatments (grazing/exclosure) at the beginning of the study and after twelve months of grazing exclusion (fig. 1). Twenty-four and 36 months since grazing exclusion, total species and forbs species richness were higher under grazing, while grass species richness was higher in

exclosures only after 24 months of grazing exclusion. In exclosures, total and forbs species richness were significantly reduced at 24 and 36 months of grazing exclusion, while grass species richness remained constant all through the experimental period (fig. 1). In grazed areas, species richness (total, grasses and forbs) declined only at 24 months since the beginning of sampling.

Bare soil percentage was not statistically different between treatments at the beginning of the study (fig. 2). At twelve, 24 and 36 months since grazing exclusion, percentage of bare soil was significantly higher in grazed areas. In exclosed areas, bare soil percentage was reduced after 12 months of herbivores removals and remains constant all through the rest of the experimental period. In grazed areas, bare soil percentage increased only at 24 months since the beginning of the study.

At the beginning of the experiment, above-ground biomass was similar in grazed areas and exclosures, while at twelve, 24 and 36 months of grazing exclusion, above-ground biomass was almost three times higher in exclosures than in grazed areas (fig. 3). In exclosures, above-ground biomass

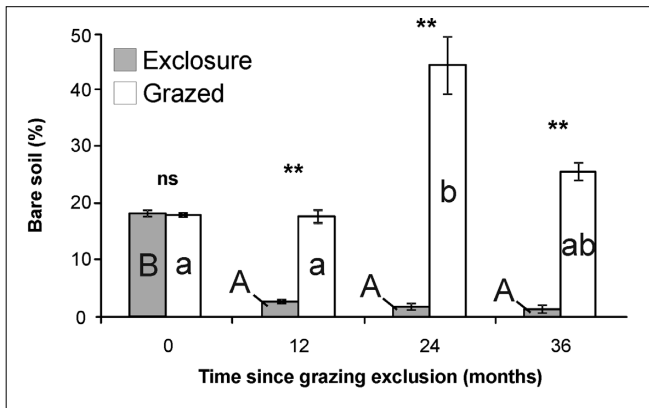


Figure 2 – Bare soil percentage in exclusions and grazed areas. ns: non-significant differences between treatments, **: highly significant differences ($p < 0.01$) between treatments. Different letters on bars are significantly different mean values (Tukey test at $p < 0.05$). Capital letters are used for non-grazed areas and lower-case letters for grazed areas. Bars represent ± 1 standard error.

increased significantly twelve months after grazing exclusion and remained constant since then, whereas in grazed areas above-ground biomass showed a significant decrease following 24 and 36 months of the experimental period.

Three- and 15-years old exclusions did not differ in species richness (total, grasses and forbs), bare soil percentage or above-ground biomass.

Changes in botanical composition

Botanical composition changed through time and in response to treatments (grazing vs. exclusion), allowing the differentiation of four groups (fig. 4). One group (Group I in fig. 4) included samples collected in 2006, at the beginning of the study. A second group (Group II in fig. 4) included all samples from the grazed areas in 2008 and 2009 (24 and 36 months since the beginning of the experimental period, respectively). A third group (Group III in fig. 4) included all the plots sampled twelve months after the establishment of the exclusions (December 2007). The last group (Group IV in fig. 4) included only exclusions sampled 24 and 36 months after the establishment of the exclusions and the plots sampled in the 15-years old exclusion. Indicator species analysis showed several species that were related to these groups (see table 1).

DISCUSSION

Montane grasslands at the Southern Pampas in Argentina showed a rapid change in species richness, bare soil percentage, above-ground biomass and botanical composition in response to exclusion from feral horses grazing. Three years after grazing exclusion, these structural variables showed a high level of recovery, as suggested by their similarity to 15-years old exclusions and by their differentiation with respect to grazed areas. Moreover, in exclusions, the botanical composition moved toward that characteristic of mature seral stages, dominated by perennial grasses such as *Piptochaetium hackelii* and *Chascolytrum subaristatum* (Frangi &

Bottino 1995). However, these results should be compared with some caution as this study refers to a short-term experiment, while the magnitudes and even the direction of change may vary considerably with the history of abandonment (Olf & Ritchie 1998). Natural grasslands do not always respond continuously to grazing and shifts toward alternative states are possible. In particular, overgrazing may lead to the creation of alternative undesirable states (Westoby et al. 1989, Friedel 1991, George et al. 1992, Cingolani et al. 2005). These transitions not only depend on grazing regime (i.e. intensity, type of herbivores, grazing management) but also may change with environmental conditions such as fire regime or extreme weather conditions, or a combination of these (Cingolani et al. 2005).

Grazing exclusion seemed to promote the decrease in species richness, particularly due to the disappearance of forb species. Comparable results have been reported for the study area (Loydi & Distel 2010) and for other similar grasslands worldwide (Pykälä 2005, Stockton et al. 2005, Altesor et al. 2006, Schultz et al. 2011). The reduction of forb species caused by grazing exclusion may be mediated by competitive suppression by grasses (Distel et al. 1996, Dickson & Busby 2009, Yelenik & Levine 2010). Taller species are more effective competitors for light and space (Anten & Hirose 1999, Grist 1999), reducing the chances of survival of smaller species (Schwinning & Weiner 1998). Species with smaller photosynthetic structures and concentration of biomass close to the ground are dominant as disturbance intensity increases (Díaz et al. 1992). In accordance with this, after three years of grazing exclusion above-ground biomass was similar to that corresponding to 15-years old exclusions and to values reported for longer term exclusions in the study area (Frangi et al. 1980). This is due to an increase in grass biomass (A Loydi, pers. observ.). Grazing by large herbivores reduces the basal diameter of grasses, but increases their density (Sala et al. 1986), aiding a rapid recovery by vegetative growth after grazing exclusion. It is worth mentioning

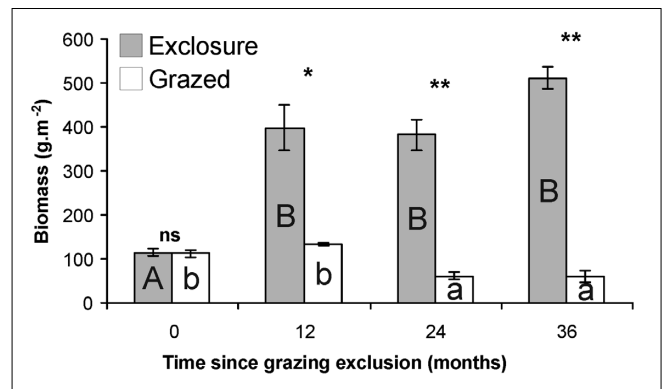


Figure 3 – Temporal changes on above-ground biomass in exclusions and grazed areas. ns: non-significant differences between treatments, *: significant differences ($p < 0.05$) between treatments, **: highly significant differences ($p < 0.01$) between treatments. Different letters on bars are significantly different mean values (Tukey test at $p < 0.05$). Capital letters are used for non-grazed areas and lower-case letters for grazed areas. Bars represent ± 1 standard error.

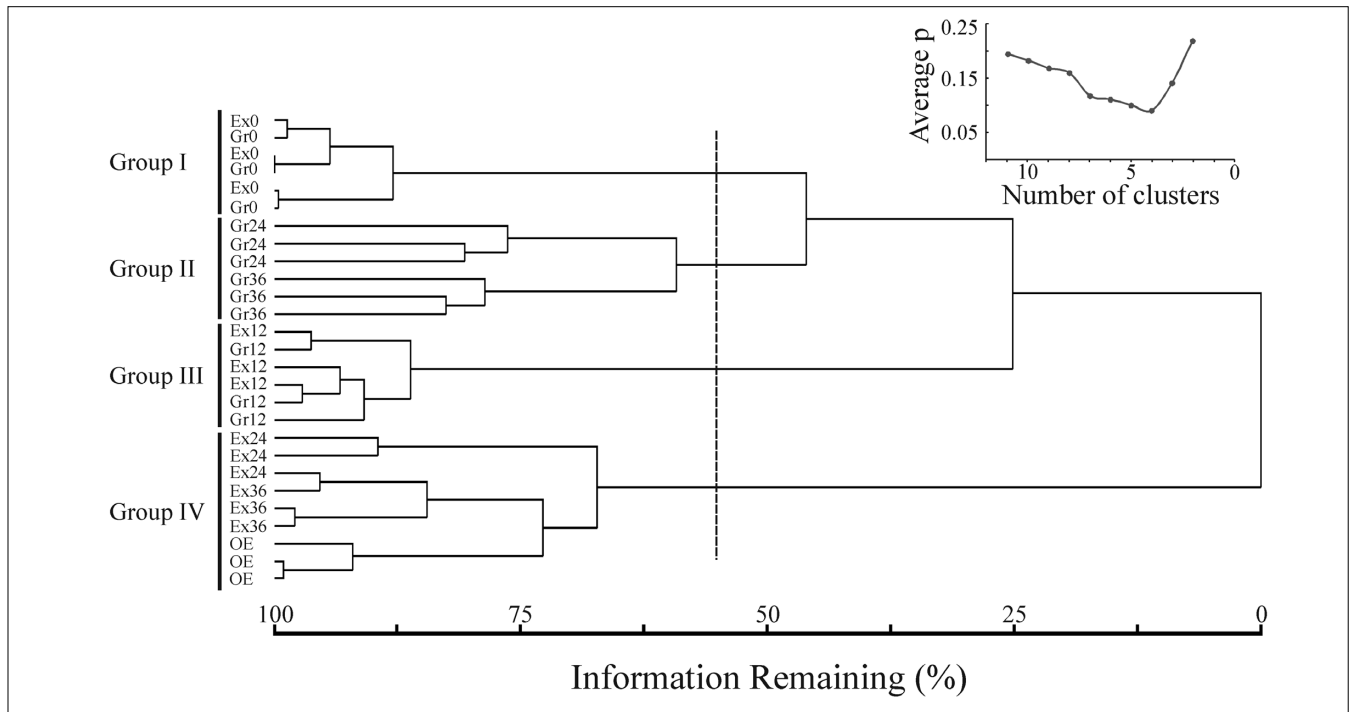


Figure 4 – Dendrogram of the cluster analysis performed for the vegetation relevés in enclosure and grazing situations. Inset figure shows averaged p-values across all species for each step of clustering. The number of clusters with the smallest average p-value was taken as pruning value (dashed line)(see Methods section for details). Ex: exclosed plots, Gr: grazed plots, OE: plots in the 15-years old enclosure. Numbers (0, 12, 24 and 36) denoted months since grazing exclusion.

that a fast recovery in above-ground biomass may also reflect a fast recovery in grassland functionality, such as productivity and nutrient cycling (Tilman et al. 1996). Nevertheless, three years is a short period of time in order to draw conclusions regarding herbivore effects on ecological functions and this deserves longer studies.

Bare soil was significantly decreased after one year of grazing exclusion. This shows that overgrazing by feral horses by removing perennial grasses and pulverizing the surface soil can increase bare soil percentage and soil erosion risk (Greene et al. 1994, Cerdà 1997, Chartier et al. 2011). This may be related with horses' morphophysiology and size. In comparison with cattle or 'guanacos', horses graze much closer to the ground (Rook et al. 2004) and are heavier making a larger impact on the structure of the vegetation and on soil surface (Loydi et al. 2010).

Botanical composition in exclosures plots showed changes toward that characteristic of late seral communities in these grasslands (Frangi & Bottino 1995), with high cover of perennial grasses. These grass species are desired by horses by their low fiber content (e.g. *Piptochaetium hackelii*), but in the presence of herbivores they virtually disappear (Loydi & Distel 2010). Conversely, plant communities in grazed areas remained stable during the experimental period, except in 2007 (Group III in fig 4). Particularly wet conditions in 2007, especially during the summer, may have facilitated annual species (e.g. *Briza minor*, *Lolium multiflorum*, *Medicago minima*, *Pelletiera verna*) and perennial hemicryptophytes (e.g. *Anemone decapetala*, *Oxalis articulata*) All these species are common in the seed bank of grazed areas

in the study area (Loydi et al. 2012). One grazing-resistant grass (*Piptochaetium montevidense*) was also characteristic of this group. It is a palatable species with the capacity to change growth form and create lawn areas with horizontal leaves and scape grazing (Loydi & Distel 2010), a similar response had been found in other grass species (Hickey 1961, Díaz et al. 1992, Fahnestock & Detling 2000). The same response is found in the grasses *Piptochaetium stipoides* that was characteristic of the grazed areas in group I. Few forb species typical of disturbed areas were also common in these grazed plots (e.g. *Geranium albicans*, *Plantago myosuroides*, *Silene gallica*). Grazed areas in group II were sampled during or immediately after the drought period (see below) and they were characterized by a subshrubs endemic species (*Mimosa rocae*) typical of xeric microsites (i.e. shallow soil) (Frangi & Bottino 1995) and two non-native forbs (*Hypochaeris radicata* and *Scleranthus annuus*). These grazed areas were also dominated by the unpalatable grass *Aristida spigazzinii* (Rúgolo de Agrasar et al. 2005), a small perennial grass with high fiber content and clonal reproduction, which allow to resist grazing.

Grasslands in the study area were subjected to moderate drought conditions during the autumns in 2008 and 2009 (data not shown), which had differential effects on grazed and non-grazed areas. Grazed areas lost above-ground biomass in both years and showed a decrease in species richness in 2008. In contrast, above-ground biomass did not change in the exclosures, which may be attributed to improved drought tolerance due to the recovery of vigor in grass species (Sala et al. 1986). On the other hand, species richness declined in

exclosures in both years (2008 and 2009) mainly due to the disappearance of forb species, a process already reported for grazing exclosures (Jacobs et al. 2007) that may have been hastened by drought conditions. Bare soil showed a 2-fold increase during the dry period in the grazed areas (see fig. 2), while in exclosed areas remained constant.

The fast recovery of vegetation in montane grasslands at the Southern Pampas in Argentina may be rooted in their evolutionary history of grazing by 'guanacos' (Barberena et al. 2009) and the high resource availability, which facilitate plant recovery after defoliation (Ferraro & Oesterheld 2002). Also, recurrent disturbance by fire and drought may have resulted in exaptations (sensu Coughenour 1985) to grazing disturbance. In this kind of systems recovery from degraded states is less dependent on the application of restoration techniques (e.g. seed bank enrichment, prescribed burning) (Bakker et al. 1996, Holmes & Cowling 1997, Pywell et al. 2002, Warren et al. 2002, Donath et al. 2007, Distel et al. 2008, Kiehl et al. 2010, Jaunatre et al. 2012). However heavier grazing and/or the presence of herbivores during more extended times may lead to the creation of alternative undesirable states (Cingolani et al. 2005). At the other extreme, grazing and fire suppression could lead to a reduction in plant diversity by competitive exclusion of dominant species.

CONCLUSIONS

Grazing intensity should be controlled in order to promote plant diversity without compromising survival of the most preferred species. Endemic species are present in areas with and without grazing, but non-native species are more common in grazing situations (see table 1). Additionally, overgrazing increased the bare soil percentage, which is related with higher erosion risks (Greene et al. 1994, Cerdà 1997), especially in montane areas. However, a short period after herbivores removal, the presence of bare soil is scarce. In natural grasslands moderate grazing intensities, different type of herbivores or different grazing management (e.g. rotational grazing) could have a positive role in reducing the risk of erosion, maintenance of ecological function and productivity, and biodiversity conservation at the same time.

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