Ecological Engineering 42 (2012) 42–53

Contents lists available at SciVerse ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Disrupting woody steady states in the Chaco region (Argentina): Responses to combined disturbance treatments

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ARTICLE INFO

Article history:
Received 15 September 2011
Received in revised form
20 December 2011
Accepted 31 January 2012

Keywords:
Roller chopping
Prescribed fire
Chaco region
Woody plant control

ABSTRACT

Vegetation states with low suitability for cattle raising ('woody states') are widespread in the Chaco region (NW Argentina). We assessed the success of roller-chopper (RR); roller-chopper with seeding of *Panicum maximum* cv green panic (RS), and roller chopper followed by prescribed fire (RF) in disrupting a woody state in two separate experiments, monitored from 1997 to 2002. We evaluated amount and temporal dynamics of woody plant volume, standing herbaceous biomass and livestock accessibility. We also monitored soil moisture at two soil depths, sunlight availability and the dynamics of germination of grass, forbs and woody species. Spatial variation was included as ecological sites, and all treatments were grazed. The longevity for each treatment was estimated by monitoring the woody volume through time. Data were analyzed using variance and regression analysis. MRPP techniques were used to study the effect on species diversity. Results were compared with information in literature to define thresholds and success of restoration. RF showed the lowest mean woody canopy ($p > F = 0.0001$), the largest mean accessibility (>70%), and longest return interval (>6 years) among treatments. RS showed the largest amount of herbaceous standing biomass (>5000 kg dry matter ha^{-1}, $p > F = 0.0001$), and a longevity >5 years. Prescribed fire was successful in controlling the increase of woody volume only at lowland and midland ecosites. At upland ecosites in RS and RR, mean woody volume and accessibility were higher than in the other two sites. Plant diversity was significantly affected by ecosite rather than by treatment and time. Soil moisture was influenced by site ($p > F = 0.001$), while sunlight availability was affected by treatment ($p > F = 0.001$). Plant germination was affected by treatment and site. The roller chopper is an adequate tool for disrupting woody states in the Chaco and could be successfully combined with prescribed fire.

1. Introduction

Grasslands, forests and savannas that have degraded to a shrub woody state are currently a major limitation for cattle operations in arid and semiarid regions of the world (Asner et al., 2004). We evaluated the success of mechanical treatments and prescribed fire in disrupting a woody state, and restoring a grassy state in the western Chaco (NW Argentina). This research was aimed at the development of management guidelines for ranchers and farmers.

At the beginning of the XX century, the vegetation of the Chaco region was a mosaic of forests, woodlands, savannas and shrublands (Bucher, 1982), suitable for livestock raising and/or timber operations. Dense shrub thickets and overstocked secondary forests are today widespread in the region. These changes in landscape and vegetation physiognomy have been caused by livestock overgrazing, indiscriminate logging, changes in the fire regime, and fencing (Adámoli et al., 1972; Boletta et al., 2006; Kunst et al., 2006). 'Woody states' in the Chaco are an undesirable, impoverished state of their local ecosystems, since their environmental and agronomical 'suitability' are low (Kunst et al., 2006). Some native species may have disappeared locally, causing a decline in biodiversity compared to the original vegetation (Bucher and Huszar, 1999). Herbaceous standing biomass is insignificant due to either low plant density and/or small plant size. The high density of woody plants and their thorny stems also decrease forage accessibility and hamper livestock and personnel movements (Nai Bregaglio et al., 1999). Current stocking rates are negligible, a fact that negatively influences the economy of ranching operations in

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doi:10.1016/j.ecoleng.2012.01.025

areas with woody states (Fumagalli et al., 1997; Kunst et al., 2006). These problems are shared with other arid and subhumid regions of the world, creating complex management situations (Gifford and Howden, 2001; Burrows, 2002; Asner et al., 2004; Hamilton and Ueckert, 2004; McIvor, 2006; Sawadogo et al., 2005; Van Auken, 2009).

Whenever a threshold in woody encroachment is attained, the limitations of grass standing crop cannot be surmounted unless a major disturbance aimed at reducing the dominance of woody plants is applied (Cabral et al., 2003; Ansley et al., 2006; McIvor, 2006; Díaz, 2007; Ansley and Wiedemann, 2008). 'Active approaches' (McIver and Starr, 2001) such as chemical products, prescribed fire and mechanical treatments have been assessed experimentally in the Chaco to disrupt woody states and to transform them into grassy states. Research reports a yield increase >500% of native grass species in comparison with untreated areas, although the efficacy in controlling woody species is variable among treatments (Rodríguez Rey et al., 1983; Alessandria et al., 1987; Galera, 1990; Passera et al., 1996; Boetto et al., 1999; Kunst et al., 2001; Casillo et al., 2006). Nowadays, 'roller chopping' is widely used in the Chaco region. Reasons are a lack of fine fuel for carrying out a prescribed fire, high cost of chemical products and ecological concerns. Roller chopping is a mechanical disturbance aimed at creating a park-like vegetation structure, which is more suitable for livestock operations than the current dense, homogeneous 'woody state' (Kunst et al., 2003). Woody individuals – especially trees – with DBH > 10–15 cm are left standing in different patterns and densities, while at the same time the rest, especially shrubs, are crushed. Forage yields and stocking rates usually increase due to the seeding of a grass species simultaneously with the roller chopping. Exotic species of the genus *Panicum (Poaceae)* are commonly used (Ledesma, 2006). They are summer growth perennials which possess the C_4 syndrome, and are adapted to shadow. They also have a high nutritional quality and yield potential; and are highly preferred by domestic livestock (Smit, 2005).

Short-term effects of roller-chopping on standing biomass of native herbaceous vegetation and on soil attributes, such as water infiltration and fertility, has been evaluated in the Chaco region (Alessandria et al., 1987; Galera, 1990; Nai Bregaglio et al., 1999; Aguilera and Steinaker, 2001; Kunst et al., 2003; Anriquez et al., 2005). However, several agronomical and ecological issues need yet to be quantitatively assessed, such as: (a) the increase of aboveground biomass yield generated by exotic grass species compared to that of native grasses and forbs; (b) effect of mechanical treatments and fire (single and combined) on the 'longevity' of the desired responses; and (c) effects of mechanical treatments with and without seeding on vegetation diversity. Native woody plants 'return' because most are sprouters: mechanical treatments and/or fire rarely kill them (Bond and Midgley, 2001; Van Auken, 2009). An important aspect in planning integrated woody plant control programs is their 'longevity', referred in literature as persistence, duration of the treatment, treatment life, and thickening (Heitschmidt and Pieper, 1983; Noble et al., 2005; Gifford and Howden, 2001). It estimates how long the 'disruption' caused by a disturbance will last. Naveh (1990, 2004) based on the Prygogyne dissipation equation, stated that the main issue of a management program is to establish the regime of disturbance, defined by the longevity (∼'return interval') of treatments. Concerns are both economical (treatment amortization, Teague et al., 2008) and ecological, since the disturbance cannot have a return interval that may endanger the sustainability of the population of a desirable woody or wildlife species (Navall, 2008). Mechanical treatments alter vegetation structure and the introduction of an exotic species may affect plant diversity (Rollins and Bryant, 1986; Bozzo et al., 1992; Nolte et al., 1994; Fensham et al., 2011), an issue that has not been studied yet in the Chaco region.

Our first two objectives were to assess changes of: (a) the aboveground biomass yield of the herbaceous stratum and (b) the canopy volume of the woody plants in a 'woody state' disrupted by roller-chopping, seeding of an exotic species and prescribed fire compared with a untreated control. By influencing the current stocking rate and hindrance for livestock and personnel, both aspects 'define' the suitability of a paddock for livestock operations. Our third objective was to assess the impact of roller chopping and seeding on the botanical composition of the vegetation, an attribute related to ecosystem biodiversity. The spatial variation (∼position in the landscape, soils), was taken into account to control this source of uncertainty on the outcome of the experiment. Additionally, effect of the treatments on soil moisture and sunlight availability was also addressed. The experiments included livestock grazing and several years of observation.

Success in pasture 'restoration' is not clearly established neither in the Chaco region nor worldwide (Zedler, 2007). In order to objectively define success it is necessary to define the 'target state', and the structural and functional goals to be achieved (Pastorok et al., 1997; Thom, 1997). One of the key functions of the Chaco ecosystem from a cattleman's point of view is to provide forage. Hence, 'success' in restoration of the grassy state could be measured by the suitability for cattle operations of vegetation physiognomy achieved; and the magnitude and duration through time of the response of: (a) herbaceous standing biomass; (b) accessibility, and (c) longevity of the treatments. Our first hypothesis was that roller-chopping and associated practices will increase standing herbaceous biomass up to 50% of the standing biomass reported in the literature of the Chaco region for native ranges in good condition. The second hypothesis was that roller chopping without seeding of introduced grasses, followed by prescribed fire will present the extended longevity among all treatments, above the 3 years threshold suggested by the local literature (Casas et al., 1978). Our third hypothesis was that roller chopping with the introduction of Green panic would increase the dominance of this species, thus reducing plant diversity of the treated areas (Williams and Baruch, 2000; Fensham et al., 2011). Finally, these findings were used to develop appropriate management guidelines of woody states in the Chaco region.

2. Materials and methods

2.1. Study site

Research was conducted at the 'La María' Experimental Ranch, Santiago del Estero Experimental Station, Instituto Nacional de Tecnología Agropecuaria, Argentina (28°03′S 64°15′E). Climate is semiarid, with 574 mm of annual average rainfall (1934–2000 series, INTA Santiago del Estero Experimental Station, 2010). Winters are cold and dry, and summers hot and rainy (Boletta, 1988). At a scale ∼1:20,000, soil and vegetation types of the study area are located along a catena from the 'upland' to the 'lowland'. The former have coarse textured soils and sustain a hardwood forest of *Aspidosperma quebracho blanco* Schlecht and *Schinopsis lorentzii* Griseb. Engl., while the latter present more developed and finer textured soils, supporting a savanna of the bunchgrass *Elionorus muticus* Spreng. Kuntze. *A. quebracho blanco* is an evergreen hardwood species that may reach a height = 20 m and a DBH = 80 cm (Tortorelli, 2009). *S. lorentzii* is a hardwood deciduous species that sheds its leaves in early-spring (October), reaching a height = 25 m and a DBH = 1.50 m. *E. muticus* is a bunchgrass 0.5 m tall (Burkart, 1969). 'Parks' are located in the 'midlands', between the other two topographic positions (Adámoli et al., 1972). This soil-vegetation catena could be interpreted as upland, midland and lowland ecological sites, respectively (Kunst et al., 2006).

Using aerial photos scale 1:20,000 and field surveys two areas which contained the three ecological sites were selected for this research. Soils of the upland and midland ecosites of the selected area were classified as Entic Haplustols, while those of the lowland were classified as Typic Haplustols (Anriquez et al., 2005). Before roller chopping, the three ecological sites were covered by a homogeneous, almost impenetrable woody strata dominated by shrub species. *Acacia aroma* Hook at Arn., and *Schinus spp*. were common in lowland sites, whereas *A. furcatispina* Burk., and *Celtispallida* Torrey in upland sites. Scattered, medium to large size individuals of *S. lorentzii*, *A. quebracho blanco*, and *Zyziphus mistol* Griseb. (DBH > 25 cm) emerged in the midland and upland sites. The herbaceous stratum was almost nonexistent. This woody state likely resulted from overgrazing, lack of fire, and drought that occurred in 1988–1989, when annual rainfall barely reached 200 mm (Kunst, personal observation).

2.2. Experimental units, treatments and experimental design

The three ecosites described above were included in each experimental unit, inasmuch as they represent the spatial variation in soil and vegetation in the Chaco region at management scales (Kunst et al., 2006). In that way, reductionism was avoided and the probability of ample conclusions increased, following Carpenter (1998) and Schindler (1998). However, increasing costs associated with logistics and infrastructure related to roller chopping, prescribed fire and grazing escalated, limiting the possibility of replicating experimental units. In order to circumvent these issues, we used a complete duplication of an unreplicated experiment. Therefore, two experiments were conducted: Exp. 1 started on January 15, 1997 and Exp. 2, on December 15, 1998, with the application of roller-chopping, completed during the following 10–15 days. The timing of roller-chopping (midrainy season) was chosen to enhance the likelihood of germination of grass species. In each experiment, five plots = 3 ha each, containing the three ecosites described above were used as experimental units. Treatments were: roller chopping with instantaneous seeding of *Panicum maximum* (Jacq.) cv Petrie (Green panic) (RS); roller-chopping without seeding (RR), and roller chopping without seeding followed by prescribed fire (RF). Two plots were used as controls (C) as a way to increase the degrees of freedom of the treatment effect in the statistical analysis (Underwood, 1998).

We defined disturbance as an event that kills or partially destroys biomass in a site, creating new after-disturbance environmental conditions (Sousa, 1984; Platt and Connell, 2003). Parameters of a disturbance such as intensity and severity can be used to predict their outcomes and facilitate the transfer of knowledge (White and Jentsch, 2001; Norkko et al., 2006; Buhk et al., 2007). Severity, interpreted here as the effect of the mechanical disturbance on the original vegetation of the plots, was described by visually assessing type and size of woody individuals crushed and amount of residues left behind.

The mechanical disturbance served several purposes: (a) to create site availability by reducing the aboveground structure of shrub and small trees (also called thinning or clearing in literature); (b) to increase sunlight availability, and (c) to facilitate forage accessibility and livestock movements. Roller chopping was aimed at controlling shrub plants, since they are the main source of 'problems' instead of trees. The top soil horizon was slightly disturbed to enhance seed germination. The roller-chopper used was an iron drum with diameter = 1.4 m and width = 2.5 m, filled with 3000 kg of water, armed with blades. In both experiments, the roller-chopper was passed twice in each plot, in oblique directions, towed by a D4 Caterpillar bulldozer.

The purpose of seeding Green panic was to increase propagule availability of a desirable species in case the seed bank of native grasses was depleted. Seeding rate of Green panic in RS was 6 kg ha$^{-1,}$ following commercial recommendations (Perez H. personal communication).

The objective of prescribed fire was to control woody resprouts. Also, it complemented the effect of mechanical treatment by eliminating plant residues limiting accessibility of personnel and livestock. Fire was applied to RF plots when the standing herbaceous biomass reached an average threshold ~2000 kg dry matter ha^{-1} of fine fuel. This happened in October 1999 for Exp. 1 and October 2000 for Exp. 2. A headfire, perpendicular to prevailing wind direction, was lit by drip torches following prescriptions of Britton et al. (1987). Fire intensity and severity were assessed using the suggestions of Ryan and Noste (1985).

In RR and RS, grazing began after one growing season rest after the initial treatment, while in RF grazing began one growing season after the burn. Plots were grazed once a year, during late winter-early spring (October–midDecember), either by mature cows and/or heifers, until the experiments were completed. Current standing herbaceous biomass was consumed to a stubble height = 15 cm, based on suggestions of Renolfi et al. (1987). Grazing system applied was rotational. Control 2 was not grazed in both experiments.

2.3. Vegetation sampling

On a yearly basis (1997–2002 for Exp. 1, 1998–2002 for Exp. 2) at the end of the growing season, the following attributes were assessed in each plot and ecosite:

2.3.1. Woody strata

(a) Mean plant density: We used the *T*-square method (Krebs, 1999), in 6 transects by treatment (2 transects by ecosite). The starting point of a specific transect was located randomly and changed every year. Transect length was 100 m, with $n = 10$ sampling points per transect. Scientific name, plant height (m) and canopy diameter (m) of each plant <3 m height was registered, using a ±0.10 m precision. Mean woody plant density (plants ha^{-1}) was calculated using formulas provided by Krebs (1999). Canopy volume (m^3) for each woody individual was estimated by assimilating its form to a known geometrical figure (e.g. semisphere, cone, etc.). Mean plant volume was calculated for each transect. Average volume of the woody strata (m^3 ha^{-1}) was estimated by multiplying mean woody density and mean woody plant volume per transect, scaling (averaging) by ecosite and plot. Botanical composition of the woody strata was estimated by calculating the species relative frequency in each transect and averaged by site. (b) *Germination of woody and broadleaf species*: It was registered only in Exp. 2, by counting the number of germinated plants in a sampling unit size = 1/10 m^2, $n = 10$, randomly distributed in each ecosite. Observations were performed every week until no germination was observed. (c) A*ccessibility*. In 2002, at the end of the experimental period, 5 imaginary planes (height = 1.5 m × length = 5 m, area = 7.5 m^2) were randomly located per ecosite in each plot. The base of the imaginary plane was identified by a tape located in the ground and limited by each side by wood poles. The height and width (m) each shrub and small tree canopy intercepting the plane was registered. Accessibility was estimated as follows

[formula (1)]:

$$\text{Accessibility}(\%) = 100 - \left(100 * \frac{\text{Area}\,[\text{m}^2]\,\text{intercepted by woody plants}}{\text{Total area}\,[\text{m}^2]\,\text{of the imaginary plane}}\right). \quad (1)$$

2.3.2. Herbaceous stratum

(a) Germination of grass species. We used the same procedure described above for woody species. Newly germinated grass plants were classified as either Green panic or native species. The first are easily distinguishable because of their reddish color in the base of the stem. Forb germination was only registered in Exp. 2. *(b) Standing biomass* (kg DM ha^{-1}) and *species relative frequency* (absence–presence, %) by using the method BOTANAL (Tothill et al., 1978), with a quadrat = 0.25 m^2 as sampling unit ($n > 52$ per plot, $n > 18$ per site). Sampling was conducted at the end of the rainfall season, prior to the grazing period. Herbage samples taken in the field were oven-dried at 60 °C for 48 h until a constant dry weight was attained. Results were expressed in kg dry matter (DM) ha^{-1}. All plant species nomenclature followed Ragonese (1951) and Burkart (1969) and was updated by consulting the 'Catálogo de Plantas Vasculares de la República Argentina' (Instituto de Botánica Darwinion, 2011).

Standing dry biomass of grass and broadleaf plants was used as an estimator of herbage biomass yield (Mitchell, 1983), and as an indirect estimator of stocking rate. Our interpretation was that the two are positively correlated.

2.4. Environmental monitoring

Air temperature (°C) and rainfall (mm) observations taken daily at the INTA Meteorological Station of the Ranch headquarters, located 7 km from the research area, were used as a 'master' record. Soil moisture was assessed gravimetrically in situ at two soil depths: 0–30 and 30–60, $n = 3$ per site, randomly located, every month during 1997–1998 for Exp. 1 (19 sampling dates) and during 1999–2001 for Exp. 2 (32 sampling dates). Results were expressed in percent, weight/weight. Measurements of sunlight availability (PAR, $\mu mol\, s^{-1}\, m^{-1}$) were taken in Exp. 1 only, at a height = 1 m, using a LI-COR bar model A-129 in treated and control plots, in 5 random points by ecosite. During 1997 – early 1998, PAR measurements were performed one day at the 2d, 3rd and 11th month after the beginning of the study, between 10 and 12 a.m. During late 1998 and 1999, PAR was assessed one day only, during the fall, spring and summer seasons, between 10 and 12 a.m.

2.5. Statistical analysis

Each experiment was analyzed separately since there were differences in the features of the treatments applied, such as the timing of roller chopping and prescribed fire, so they were considered duplicates. The effects of treatments, as related with our hypotheses on the standing dry herbaceous biomass and woody canopy volume, were tested using an analysis of variance with a repeated measures approach (Littell et al., 1998). Field data of herbaceous biomass and woody volume gathered in the transects were scaled (averaged) by plot and ecosite and used as dependent variables. Independent factors were treatment proper, ecosite, 'years since initial treatment', and their interactions. 'Treatment' represented roller chopping, seeding, and prescribed fire effects upon dependent variables at the 'landscape level' since the data were averaged by plot. 'Ecosite' was considered as a classification factor representing the combined effects of topographical position, water runoff–runon phenomenon, soil texture, and introduced the spatial variation at scale < 1:20,000. Ecosites were considered nested within treatments. 'Years since initial treatment' was calculated by summing up the number of days that passed between the day on which treatment began until the day of each sampling observation, divided by 365 days. In this step of the analysis, 'years since initial treatment' was considered a class variable. It represented the confused effect of several factors: grazing and weather features during a specific year – especially rainfall amount and distribution – and the spatial variation resulting from the changing locations of transects. Although we were aware that if statistically significant interactions (effect modification) among main factors would mean that treatment effects were 'moderated' by soil, climate and grazing factors, we were mainly concerned about research questions associated to the main factor 'treatment' and our discussion and conclusions sections will be centered on it (University of England, School of Psychology, Web Stat 2011). Ecosite and 'years since initial treatment' were included in the analysis only as a means to separate sources of variance in the statistical analysis.

The hypothesis about longevity of the treatments was assessed by a separate regression analysis of the two experiments using the mean woody canopy of each plot observed each year as dependent variable, and 'years since initial treatment' as calculated in the above paragraph as an independent, continuous variable representing time as a chronological factor. The regression coefficients (β) of the resulting equations could be interpreted as an estimation of the 'growth rate' of woody plant volume through time. Our interpretation was that the magnitude, differences and sign of β among treatments will give an insight of their respective 'longevity'. The smaller the β, the larger the period between treatments needed to re-control the woody canopy.

Data of soil moisture were sorted and scaled (averaged) by treatment and were plotted versus time to visually assess their temporal dynamics. Effects of treatment, ecosite, and years since initial treatment on soil moisture were analyzed using variance analysis. Treatments considered were controls and 'roller chopping' only, since previous analysis indicated that there were no significant differences among RF, RS and RR. Analysis of variance was also performed for the data on accessibility, and germination of Green panic, native grass, broadleaf and woody species, considering treatment as independent factors. PAR was used as a dependent variable in an analysis of variance considering treatment, ecosite, and 'sampling date' in 1997–1998, and 'season' in late 1998–99 as independent variables.

All dependent variables were rank transformed previous to the statistical analysis in order to satisfy assumptions of linearity and homogeneity of variances (Conover, 1980). Calculations for the statistical analysis were performed using the MIXED, GLM and NESTED procedures of the SAS package (SAS, 2002; McDonald, 2009). The probability of obtaining a small *F* test magnitude was used for identifying statistical significant effects, considering the type III sum of squares. Statistically non significant effects and interactions were removed from the analysis. Means were compared using the Duncan test (SAS, 2002). The LSMEANS statement of the PROC GLM and MIXED was also used to calculate and compare means; however, when there were no differences among means using both tests, only the results of the Duncan test were reported.

The PROC GLM was used to calculate the regression coefficients (β) of woody canopy versus time. Comparison of β among treatments and ecosites was performed using the 'contrast' option of the PROC GLM (UCLA, 2010).

Differences among mean soil moisture and mean PAR were tested using a *t*-test. An $\alpha = 0.05$ was used for all mean comparisons.

Since treatments (=plots) were not replicated, variation comes from pseudoreplications (Hurlbert, 2004), considered a lower level

Table 1
Grazing periods and number of grazing animals in two roller-chopping experiments. 'La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station.

Experiment	Grazing period	Total grazing days	Number of animals	Category	Liveweight (kg)
1	19/08/97 to 5/09/97	15	39	Cows	300–350
	9/09/98 to 15/09/98	6	47	Cows	300–350
	21/08/99 to 10/09/99	30	49	Cows	300–350
	16/10/00 to 31/10/00	15	37	Heifers	250–300
	7/08/01 to 15/09/01	38	39	Heifers	250–300
	15/08/02 to 20/09/02	35	59	Cows	300–350
2	15/11/99 to 29/12/99	44	53	Heifers	250–300
	15/10/00 to 6/12/00	51	83	Cows	300–350
	3/11/01 to 18/11/01	15	73	Cows	300–350

of an ordinary replication (Johnson, 2006). The estimation of the experimental error of the dependent variables comes from two sources of variation: transects and/or sampling points (spatial), and sampling dates (temporal). Analysis considering sampling dates was conducted using ecosites within treatments as the source of variation.

Our third hypothesis was related to community responses to disturbances (changes in species composition and richness, i.e. diversity). The multi-response permutation procedure (MRPP) was used to assess similarities in botanical composition (frequency, %, of woody, forb and grass species) using 'years since initial treatment', treatment, and ecosites as grouping factors. A distance matrix was first calculated using the Bray-Curtis coefficient because frequency data were used (Bray and Curtis, 1957; Mc Cune and Grace, 2002). Previous to the MRPP analysis, this similarity matrix was rank transformed (Mc Cune and Grace, 2002). Subsequently, the weighted mean within group distance (δ) was calculated for the selected groups. The third step was the calculation of the difference between the observed and expected δ for each group, using T [formula (2)]:

$$T = \frac{\text{observed}\,\delta - \text{expected}\,d}{\text{sd expected}\,\delta}; \qquad (2)$$

where sd = standard deviation. Smaller values of δ indicate a tight clustering of groups, while larger, negative magnitudes of T suggest a strong separation among groups (Mc Cune and Grace, 2002). The within-group homogeneity is assessed by A, the chance corrected within-group agreement. If A = 1, all items within a group (treatments, ecosites, etc.) are identical. The PCORD package (McCune, 2002) was used for MRPP calculations and testing. Interpretation of results of MRPP requires additional data exploration to verify which species are contributing to differences in the assessed communities. Those results will be reported elsewhere.

The lay-out of this study would allow the discussion of results at different perception levels: (1) 'plot level', that considers treatment effect, concealing spatial variation by averaging field data across ecosites; (2) the 'ecosite' proper, encompassing the local variation within a specific landscape unit; and (3) the effect of interactions among main factors. The first would be important for developing management recommendations of a paddock containing one or more ecosites, while the second approach would be relevant from an ecological point of view. The latter would be relevant in assessing weather fluctuations and grazing effects on treatments. Due to the amount of information, ecosite and interactions effects will be presented at length elsewhere.

3. Results

The roller chopper crushed almost all brush and tree plants height <3 m, while trees with DBH > 10–15 cm and tall shrub plants were left standing. In upland and midland ecosites, where most of the large tree individuals were located, the resulting plant physiognomy was parkland. A grassland was 'created' at lowland ecosites, because of a natural absence of trees. Crushed woody plants resprouted abundantly, since very few of them were uprooted and killed by the mechanical treatment. Most woody residues left behind by the roller-chopper had a diameter <3 cm, and they degraded naturally almost completely after the first rainy season. The final density of standing woody plants in the midland and upland sites achieved was 500–600 woody individuals ha^{-1} (trees and shrubs) on average, considering all plant sizes. Sustained by the availability of fine fuels, fire propagated successfully only in lowland and midland ecosites within the RF plots. Average flame length was 2.5–3 m. Although fuel as consumed to the ground, soil did not change color and ashes were black. Grazing information is presented in Table 1.

3.1. Woody plant volume

Factors 'treatment' and 'years since' 'initial treatment' were statistically significant in both experiments ($p > F = 0.0001$). Ecosite was statistically significant in Exp. 1 and Exp. 2 ($p > F = 0.0353$ and $p > F = 0.096$, respectively). At the end of the experimental period (2002), mean woody canopy volume of roller-chopped plots was significantly smaller than control plots in both Exp. 1 and Exp. 2 (Table 2). RF presented the smallest mean woody canopy volume, but was significantly different from RS and RR only in Exp. 2 (Table 2).

3.2. Growth of the woody volume through time

The regression coefficients (β) of the roller-chopped plots differed significantly from zero, and were positive in both experiments (Table 3A). On the other hand, the β's of the control plots were either of very small magnitude, or did not differ significantly from zero. Among roller-chopped plots, RF presented the smallest β (Table 3A and Fig. 1). The hypothesis of equal β among treatments was rejected in Exp. 1 ($p > F = 0.0064$), but accepted in Exp. 2 ($p > F = 0.21$). The β's of the control plots were of smaller magnitude than those of the roller chopped plots, but in the Control 2 of Exp. 2. (Table 3B). In Exp. 1, the slope of RF differed significantly from RS only (Table 3B). In Exp. 2, slopes of the roller chopped plots did not differ significantly among themselves.

3.3. Accessibility

Treatment effect was statistically significant in Exp. 1 ($p > F = 0.009$) while not significant in Exp. 2 ($p > F = 0.21$). RF in Exp. 1 and Exp. 2. showed the highest mean accessibility among all treatments (Table 4). In both experiments the lowland ecosite presented the highest accessibility in RF, but in the RS and RR treatments the situation was the opposite (Fig. 2).

Table 2
Mean woody volume ($m^3\ ha^{-1}$) observed in two roller-chopping experiments. 'La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. References: RS: roller chopping with instantaneous seeding of *Panicum maximum* cv Green panic; RR: roller chopping without seeding; RF: roller chopping followed by prescribed fire. n = number of pseudoreplicates.[a] Standard deviation, between brackets.

Treatments	Exp. 1 (1997–2002)	n	b	Exp. 2 (1999–2002)	n	b
Control 2	16,997 (14,179)	9	a	28,431 (16,369)	9	a
Control 1	7985 (6908)	15	a	8894 (8700)	9	b
RR	1079 (1292)	18	b	1606 (1182)	9	c
RS	1068 (858)	18	b	1057 (419)	9	d
RF	835 (592)	18	b	627 (420)	9	e

[a] Number of pseudoreplicates: *Exp. 1*: for RR, RS and RF = 6 sampling dates * 3 ecosites. For Control 1 = 5 sampling dates * 3 ecosites, and Control 2 = 3 sampling dates * 3 ecosites. *Exp. 2*: 3 sampling dates * 3 ecosites.
[b] Within an experiment, rows followed by the same letter are not significantly different, Duncan test, $\alpha = 0.05$.

Table 3
Regression coefficients (β) of the mean woody volume ($m^3\ ha^{-1}$, rank transformed) for each treatment associated to 'years since initial treatment' (A) and comparison of slopes among treatments (B). La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. References see Table 2.

(A)

Experiment	Treatment	β	$p > \|t\| H_0: \beta = 0$
1 (1997–2002)	RS	8.61	0.0001
	RR	8.02	0.0014
	RF	4.91	0.03
	Control 1	1.82	0.15
	Control 2	−2.54	0.29
2 (1999–2002)	RS	5.4	0.06
	RR	3.87	0.08
	RF	4.8	0.01
	Control 1	0.41	0.83
	Control 2	4	0.01

(B)

	Exp. 1 (1997–2002)	Exp. 2 (1999–2002)
Slope comparison among treatments throughout time:		$(p > F)$
Control versus RS	0.0008	0.24
Control versus RR	0.0017	0.15
Control versus RF	0.0423	0.44
RS versus RR	0.78	0.81
RS versus RF	0.0945	0.71
RR versus RF	0.16	0.54

3.4. Grass and forbs standing biomass

'Treatment', Ecosite and 'Years since initial treatment' were statistically significant in both experiments ($p > F = 0.0001$). Roller-chopper plots showed a higher mean grass and forb standing biomass than controls during the study period (Table 5). RS presented the largest standing herbaceous biomass, achieving a magnitude > 5000 kg DM ha^{-1} in some years (Table 5). The mean proportion of desirable herbaceous species ('forage') by cattle was ~70%.

Table 4
Mean accessibility (%) observed in two roller chopper experiments, estimated at the end of the study period. La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. For references see Table 2. Standard deviations, between brackets.

Treatments	Exp. 1 (1997–2002)	n[a]	b	Exp. 2 (1999–2002)	n[a]	b
RF	75 (20)	15	a	93 (8)	15	a
RS	58 (27)	15	ab	92 (8)	15	a
RR	41 (35)	15	bc	87 (11)	15	a
Control 2	32 (36)	15	c	71 (26)	15	c
Control 1	25 (20)	15	c	62 (32)	15	c

[a] Number of pseudoreplicates within an experiment = 5 sampling stations * 3 ecosites.
[b] Within an experiment, rows followed by different letters indicate significant differences, Duncan test, $\alpha = 0.05$.

3.5. Environmental dynamics

Annual rainfall was near or above average during the study period (Fig. 3). Soil moisture dynamics followed the seasonal rainfall pattern. The roller-chopper treatments showed an increase of mean soil moisture of the upper soil horizon immediately after the treatment when compared to control plots and deep soil horizons. However, this increase was not statistically significant, and only lasted ~2 weeks after the treatments were applied. Afterwards, the temporal pattern of soil moisture was similar for both controls and treated plots. The ecosite effect on soil moisture was statistically significant in both experiments ($p > F = 0.0946$ for Exp. 1 and $p > F = 0.0414$ for Exp. 2, respectively), despite the large temporal and spatial variation considered. Lowland ecosites presented the

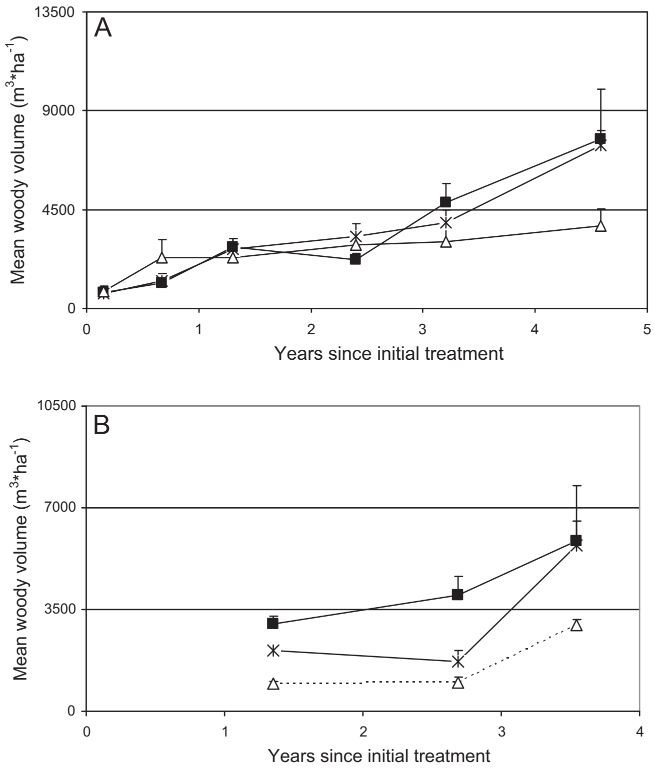

Fig. 1. Temporal dynamics of the mean woody volume ($m^3\ ha^{-1}$) through time in two roller-chopping experiments. 'La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. (A) Exp. 1 and (B) Exp. 2. References of treatments: RF: open triangle, RR: filled square, RS: asterisk. For other references see text.

Table 5
Dynamics of the mean standing herbaceous biomass (kg DM ha^{-1}) observed in two roller-chopper experiments from 1997 to 2002. La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. For references see Table 2. Control 2 was not grazed. Ecosites within treatments (=plots) were used as pseudoreplications in each sampling date.

Experiments	Date	RS	RR	RF	Control 1	Control 2
1	September-97	692	936	512	–	–
	December-97	1264	920	910	876	–
	February-98	5982	3457	3400	942	383
	July-98	741	1121	2492	1271	228
	December-98	1209	1057	1066	1044	–
	August-99	2064	3162	1825	2476	1300
	March-00	4350	2577	3550	2743	1839
	June-01	5098	4146	2887	1779	1501
	May-02	1615	965	1023	767	504
	Mean[a]	2557a	2038ab	1963ab	1487b	959c
% increase over the mean of Control 2		167	112	105	55	100
2	February-99	2948	3132	2298	2117	–
	May-00	3527	2216	2361	2254	2128
	May-01	8490	4795	4399	2239	2235
	Mean[a]	4988a	3381ab	3019b	2203b	2181b
% increase over the mean of Control 1		126.43	53.47	37.04	100.00	100

[a] Within an experiment, columns followed by the different letters are significantly different, Duncan test, $\alpha = 0.05$.

highest mean soil moisture when compared to the other ecosites, irrespective of treatment (Fig. 4).

Sunlight availability was significantly influenced by sampling date and treatment ($p > F < 0.0001$). During 1997-early 1998, mean sunlight availability in the roller chopped plots was significantly higher than in the control plots: 930.83 μmol s^{-1} m^{-1} versus 631 μmol s^{-1} m^{-1}, respectively ($p > |t| < 0.0001$). Differences among ecosites were not significant. During 1998–1999, treatment and seasonal effects were significant ($p > F = 0.0001$). The mean sunlight availability was 461 μmol s^{-1} m^{-1} and 329 μmol s^{-1} m^{-1} ($p > |t| < 0.0002$) in the treated and control plots, respectively. The mean sunlight availability of roller-chopped plots represent a ~40% to ~55% increase over the mean sunlight availability of the controls.

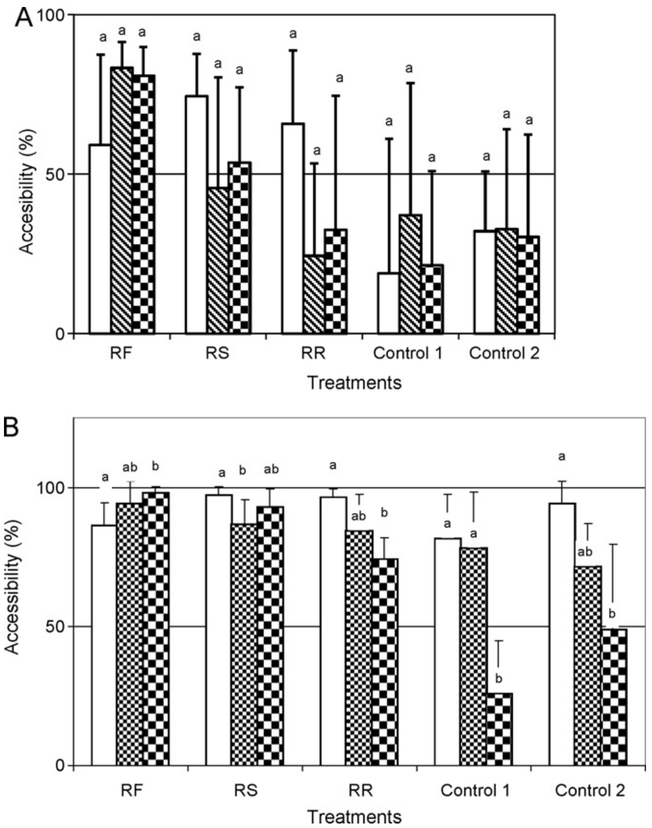

Fig. 2. Mean accessibility (%) estimated at the end of the experiments, sorted by treatment and range sites. A: Exp. 1 and B: Exp. 2. White bar: upland site; shaded bar: midland site; checkered bar: lowland site. For other references, see Table 2. Within a treatment, different letters indicate significant differences, Duncan test, $\alpha = 0.05$. Error bars represent spatial variation within a treatment, 5 sampling stations in each ecosite.

3.6. Herbaceous and woody germination

Native grass species presented a higher mean germination in roller-chopped plots than in controls in both experiments ($p > F < 0.001$, Fig. 5A). In Exp. 2, germination of broadleaf herbaceous and woody species was also higher for the roller-chopped plots than for the controls ($p > F < 0.001$ and $p > F = 0.0312$, respectively, Fig. 5B). Nevertheless, germination of both life forms was observed only in the first season after application of roller-chopping. Green panic showed the highest mean germination in the upland ecosite in both experiments (Fig. 6A), while the largest mean germination of native grass species in both experiments was observed in the midland and lowland sites (Fig. 6C).

The recruitment of new individuals by germination was relevant only in the case of native grasses: forb species changed the botanical composition for a while, but after two growing seasons, the

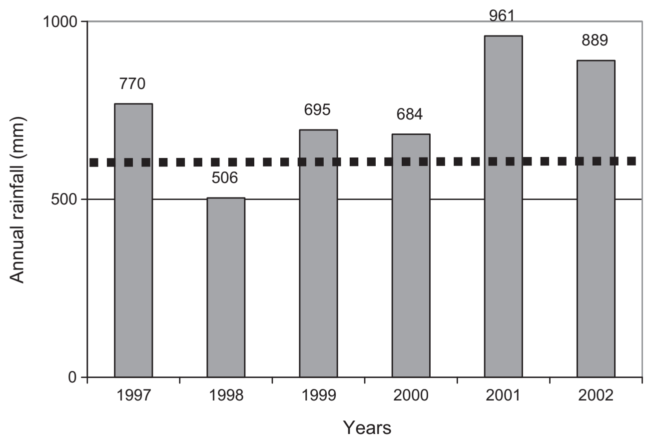

Fig. 3. Total annual rainfall between 1997 and 2002 at 'La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. Broken line represents long term average, 574 mm, 1934–2008 series.

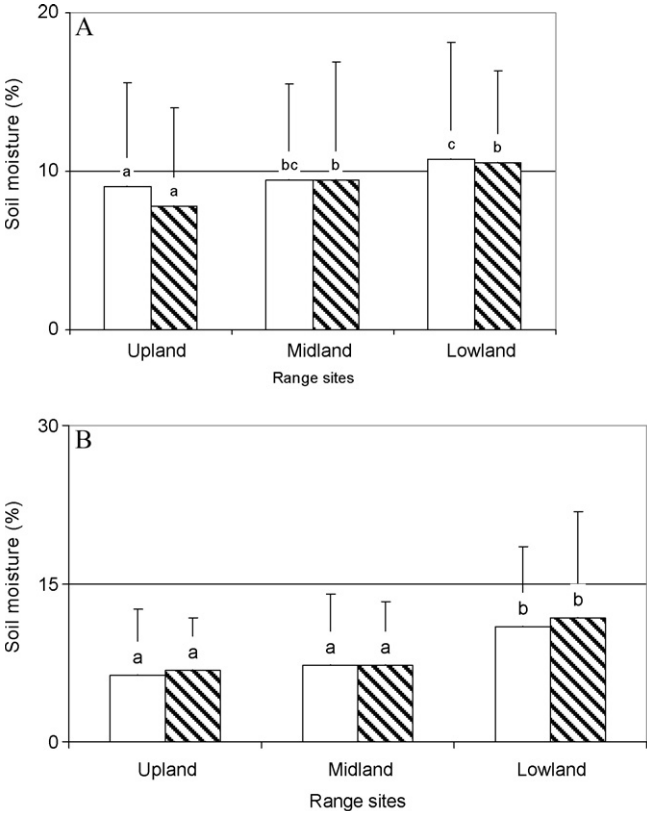

Fig. 4. Mean soil moisture (%) observed in roller-chopped plots and controls sorted by ecosite, at 0–30 cm soil depth. (A) Exp. 1, during 1997–1998 and (B) Exp. 2, during 1998–2001. 'La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. References: white bars, controls; shaded bars, roller-chopped treatments. Different letters within ecosites indicate significant differences among treatments, LS MEANS test. Sources of variation are spatial, 3 sampling points per date; and temporal, 19 sampling dates for Exp. 1 and 32 sampling dates for Exp. 2.

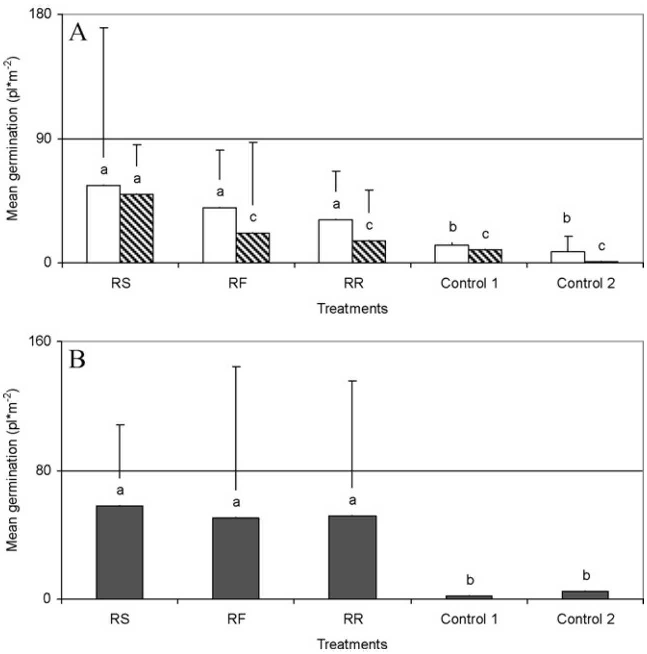

Fig. 5. Mean germination of: (A) native grass species in Exp. 1 and Exp. 2 and (B) herbaceous broadleaf species in Exp. 2. 'La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. References in A: white bar, Exp.1, shaded bar, Exp. 2. Different letters within a treatment indicate significant differences, LS MEANS test. Other references see Table 2.

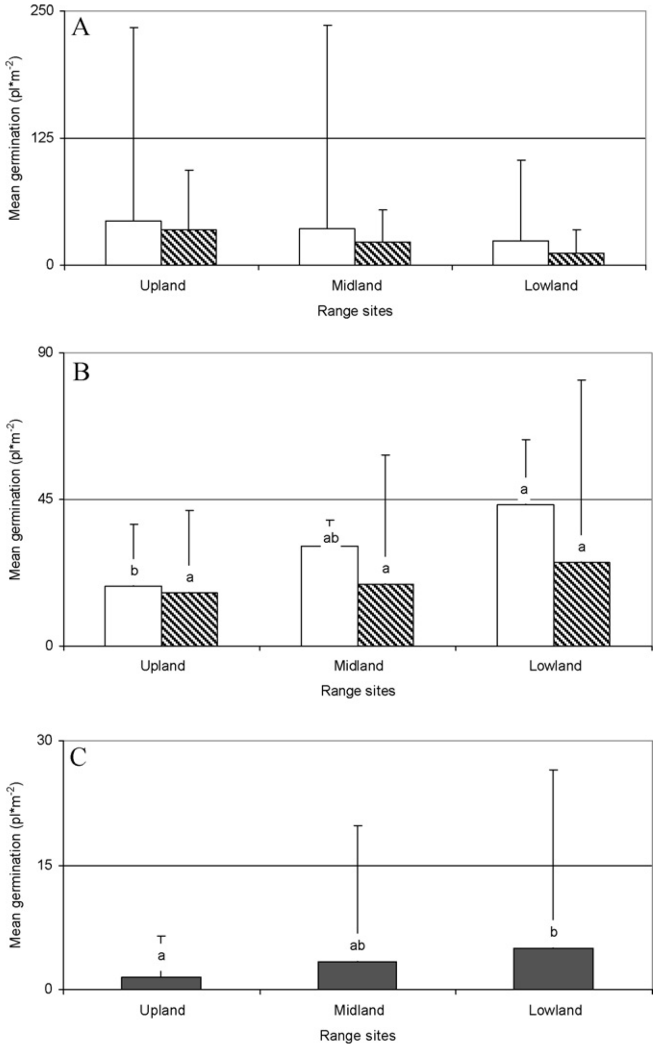

Fig. 6. Mean germination of grass species in two roller chopper experiments sorted by experiments and sites. 'La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. (A) Green panic in RS, (B) native grass species, and (C) woody species in Exp. 2. References A and B: white bar, Exp.1, shaded bar, Exp. 2. Other references see Table 2.

herbaceous stratum was dominated by grasses. Few small plants of woody species survived up to the second season.

3.7. Changes in botanical composition

Differences among observed and expected δ were larger and statistically significant when species frequencies were grouped according to 'ecosite' rather than by 'years since initial treatment' (time) and 'treatment', in both experiments (Table 6). The magnitudes of T and A also support the idea that differences in species composition for shrub, forb and grass strata are strongly influenced by ecosite (Table 6).

4. Discussion

The mechanical disturbance applied was considered to be medium in intensity and medium in severity. This characterization was based on the following attributes: (a) density of woody individuals left standing and (b) amount and size of woody residues left behind in roller-chopped plots. Fire could be characterized as being of high intensity/medium-high severity at lowland sites, but

Table 6
Result of multiresponse permutation procedures (MRPP) for Exp. 1 and Exp. 2, woody, grass and forbs species. 'La Maria' Experimental Ranch, INTA Santiago del Estero Experimental Station. References: δ = weighted mean within-group distance, T = (observed δ–expected δ/sd expected); A = chance-corrected within-group agreement. Since the distance matrix was rank transformed all expected δ = 0.5.

Experiment	Group	Factor	δ Observed	T	A
1	Woody species	Treatment	0.44[a]	−5.67	0.1135
		Site	0.39[a]	−13.66	0.22
		Time	0.48[a]	−2.02	0.047
	Grass and forbs	Treatment	0.41[a]	−9.33	0.17
		Site	0.38[a]	−16.55	0.27
		Time	0.48[a]	−1.9	0.045
2	Woody species	Treatment	0.48[a]	−1.91	0.044
		Site	0.29[a]	−25.93	0.42
		Time	0.51	0.84	−0.02
	Grass and forbs	Treatment	0.47[a]	−1.76	0.06
		Site	0.32[a]	−14.46	0.36
		Time	0.49	−0.60	0.014

[a] Probability of a smaller or equal δ < 0.05.

of medium intensity and severity at midland sites, respectively. Stocking rates could be characterized as conservative.

The amount and dynamics throughout time of mean herbaceous standing biomass and mean woody canopy volume suggest that roller chopping disrupted the initial stable woody state. Germination of grass and forb species was triggered by the mechanical disturbance, and was likely enhanced by the sudden increase of sunlight and soil moisture availability. Similar outcomes have been reported in literature for other ecosystems worldwide when mechanical treatments are used for range improvement (Rollins and Bryant, 1986; Alessandria et al., 1987; Tanner et al., 1988; Galera, 1990; Bozzo et al., 1992; Sawadogo et al., 2005; Bates et al., 2005; Ansley et al., 2006; Willcox and Giuliano, 2010).

The plant physiognomy (vegetation structure) achieved at the ecosite level of treated plots in our experiments (grassland in lowland, woodland at upland and midland) presented a more dominant herbaceous stratum than the controls and is more 'friendly' for livestock management since permits free ranging of animals and personnel.

Past research suggest that an acceptable amount of the herbaceous standing biomass for native vegetation of the Chaco region in regular-good range condition would be 2500–3000 kg DM ha^{-1}, averaging the three ecological sites (Morello and Saravia Toledo, 1959; Anderson et al., 1980; Kunst et al., 2006; Díaz, 2007). Considering native grass and forb species only, this 'threshold' was reached 1.5–2 growing seasons after initial treatment in the RR and RF plots in both experiments. It was reached even faster, in one growing season, with the introduction of Green panic (RS plot). Compared with controls, the proportional increase in standing herbaceous biomass attributed to the disturbance treatments was 160% in RS and 100% in RR and RF, respectively, considering the whole study period (Table 5). Green panic represented an average proportion of 20–30% of the standing biomass throughout the entire sampling period in RS. The herbaceous response was still 'acceptable' after 6 years of the initial treatment in RS, although Green panic only persisted in the upland ecosite. Reports from other ecosystems indicate that the period of time until grass standing crop achieves an 'acceptable' level is >3–4 years (Bates et al., 2005; Moore et al., 2006; Moustakas et al., 2009). The shorter period observed in this research is probably the result of the high fertility of Chaco soils. RS presented the largest mean standing biomass in both experiments. This fact is due to the success of Green panic in establishing in the upland site, under the shade of the woody individuals left standing.

The re-sprouting 'abilities' of woody species cause an unavoidable 'renewal', called 'thickening' in literature (Gifford and Howden, 2001). Despite this growth of woody plants, average accessibility was significantly higher in roller chopped plots than in control plots at the end of both experiments. This result was especially relevant in RF, where a combined treatment was used. Thus, the suitability of roller-chopping and fire in restoring a useful function for livestock operations of the Chaco ecosystems is confirmed. A rather astonishing result is that in RR and RS mean accessibility in the upland was similar, or higher than in the lowland and midland ecosites. This fact could be attributed to the presence of large trees not crushed by the roller-chopper. Live, mature trees left standing may control brush re-growth by limiting sunlight and competing for water at deep soil depths with shrub species. This 'positive' effect of large trees in sustaining the suitability for livestock operations has been reported for ecosystems that possess tree species with potential to develop large trunks and canopies (McIvor, 2006). The high accessibility at the upland ecosite in RS after 5 years of the initial treatment application could be also attributed to the presence of Green panic, which may compete for some resources with shrub plants.

Treatment longevity could be defined as the number of years between treatments aimed at maintaining the grassy state. It is an estimation of the disturbance regime to be applied to a paddock. In our research, 'longevity' was estimated by the magnitude of the growth rate of the woody volume through time (β). The magnitude of β is negatively correlated with treatment longevity. However, there are some intricacies that need to be clarified first. Lack of difference from zero of the β coefficients in control plots suggest that their woody canopy was not 'growing' through time, or doing it very slow. This result support the concept that the woody strata in these plots had reached a 'steady' state, as reported for other ecosystems (Ansley and Rasmussen, 2005). Therefore, an initial disturbance – such as roller-chopping – was needed to 'break' the stagnation of the woody component of the vegetation. The significant difference from zero of β of roller-chopped plots indicated that the disturbance applied triggered a 'renewal' of the woody canopy. This fact contradicts the objective of controlling the woody stratum stated for mechanical treatment. The mean woody volume of control plots showed an average amount of 16,000–28,000 m^3 ha^{-1}, not changing significantly through time. Therefore, it could be used as a practical, rough threshold to determine the period of time required for repeating a brush control treatment. In our research, the magnitudes of β of the roller-chopped plots suggest that the time needed to reach that 'threshold' by the woody stratum would be likely ≥5 years. Specifically for the central-eastern Chaco region, Casas et al. (1978) reported an average longevity of 3 years for roller chopper treatments. A partial, non-exhaustive review of scientific literature from a variety of ecosystems and management approaches indicate a 'mean longevity' ranging from 2 to 3 years for the roller-chopping treatment and 2–5 years for prescribed burning; up to a threshold of 20 years for the same treatments (Tanner et al., 1988; Bozzo et al., 1992; Welch, 2000; Bates et al., 2000, 2005; Watts et al., 2006; Moore et al., 2006; Teague et al., 2008; Moustakas et al., 2009; Van Auken, 2009; Willcox and Giuliano, 2010; Menges and Gordon, 2010). A minimum 'threshold' of acceptable longevity for mechanical disturbance would be 3 years. Based on the above paragraphs, the second part of our hypothesis could be accepted: roller chopping restored the suitability for livestock operations of a woody state, both structurally and functionally.

Our second hypothesis refers to the success and effectiveness of prescribed fire as a complementary treatment of roller-chopping. The attributes used to accept/reject this hypothesis were β and mean accessibility. RF presented in both treatments the smallest magnitude of β, indicating a return interval >6–7 years, and a mean accessibility > 60–70% at the end of the study period. These results suggest that the second hypothesis should be accepted, confirming the effectiveness of combined treatments for controlling woody plants for the Chaco region, as has been acknowledged worldwide (González and Látigo, 1981; Wolters, 1981; Henkin et al., 1998; Hamilton and Ueckert, 2004; Smit, 2005; Ansley et al., 2006; Ansley and Wiedemann, 2008). However, because of the quantity of fine fuel present, fire was successful only in the mid and lowland ecosites.

Our third hypothesis was related to changes in plant diversity. Changes in botanical composition could be due to: (a) modification of resource availability, such as sunlight, water, and nitrogen, resulting from a sudden destruction of the woody plant canopy (Bates et al., 2005) and (b) the inclusion of propagules of green panic, an exotic species. The MRPP analysis did not suggested large changes in botanical composition through time caused by the disturbances applied. In our experiments, botanical composition was significantly influenced by ecological site type – an inherent feature of the landscape at large levels of perception – rather than by treatment. Results indicated that there was a change in the dominance through time of functional groups/life forms, instead of a shift in the botanical composition. This overwhelming effect of spatial variation (topography, soils) in botanical composition and plant succession rather than management practices has been stated in literature (Johnson and Miyanishi, 2008; Fensham et al., 2011).

The Chaco region is a semiarid environment where evaporative demands usually exceed rainfall (Boletta, 1988). Water availability should be the limiting natural resource acting as a driver of vegetation dynamics, as it as been suggested for other semiarid–arid ecosystems. An important pathway of water transpiration is the canopy of woody plants. Its sudden 'disappearance' caused by disturbance may temporarily alter the water balance, thus making it more available for other functional groups/life forms (Smit, 2004; Bates et al., 2005). In our research, temporal dynamics of soil water content suggested that its availability was influenced by topographic position and resulting water runoff-runon process – 'ecological site types' – rather than by treatment, despite the large temporal and spatial variation present in the data. The different botanical composition and vegetation structure of ecological sites of the Chaco as reported by Kunst et al. (2006) may be the reflection of this fact. On the contrary, sunlight availability was significantly affected by treatment rather than by ecosite. Native grass species respond readily to the destruction of the woody canopy by fire (Morello and Saravia Toledo, 1959), suggesting that they are not adapted to shade. Therefore, sunlight availability could be the probable driver of the changes in the proportion of life forms.

5. Conclusions and management implications

According to Pretzsch (2007), the scientific knowledge needed to transform the current state of a vegetation type into a desired state has two components: (a) the '*target knowledge*'; i.e. the 'appropriate' plant structure/physiognomy (proportion of grass–woody strata) to attain an acceptable livestock operation in the Chaco (which component to clear?) and (b) the '*transformation knowledge*': the practical rules/recommendations developed for the operational realization of the desired transformation (how much to clear?, how to clear it?, indicators of success?, etc.). The target vegetation physiognomy or structure for a roller chopping disturbance in the Chaco region should be a parkland (with some scattered trees and shrubs) at low and midland ecosites, and an open woodland/forest in upland ecosites. This recommendation agrees with others made in literature, suggesting that thinning/clearing of the woody component of the vegetation should not be extreme and needs to be cautiously approached. On one hand, the restoration of the original savannas and grasslands would be an unrealistic proposal due to high costs involved. On the other hand, woody plants provide ecological services such as shade, forage (fruits and leaves), soil stability and habitat for wildlife. They also are a source of biodiversity and carbon storage (Gifford and Howden, 2001; Ansley and Rasmussen, 2005; McIvor, 2006; Navall, 2008; Le Brocque et al., 2009).

Our research indicates that roller chopping is an appropriate tool to disrupt woody states in the Chaco region. The following recommendations for its implementation are suggested: (a) treatment should be aimed at crushing the shrub component rather than the tree component of the woody stratum and (b) the disturbance intensity for roller chopping should be 'medium'. These 'prescriptions' are aimed at creating site availability, modifying species performance by increasing sunlight availability, and to allow for a competitive advantage to tree species rather than shrub species. An 'artificial' load increase of medium and heavy fuels for future prescribed fires is also avoided. A large size roller-chopper should be prevented, since it reduces selectivity (which component to clear?), and the possibility of managing the 'intensity' of mechanical disturbance. Severity of roller chopping could be estimated by the density of trees and large shrub plants left standing: a DBH > 10–15 cm could be a practical reference. Since most of the shrub species have stems with a diameter <3 cm, severity of roller chopping could also be assessed by the amount of woody residues of that size left behind and included in the fine and medium fuel categories. Use of Green panic to increase propagule availability is recommended, since the seed bank of native species may be depleted. These prescriptions are aimed at reaching the 2500–3000 kg DM ha^{-1} threshold of standing herbaceous biomass as soon as possible, and to increase the treatment longevity up to 5–6 years. The longevity observed represents a 100% increase over the 3 years threshold reported in literature. After treatment, stocking rates should be conservative.

Restoration/reclamation research is notoriously site specific: spatial variation precludes standard designs (Holl, 2010). However, that information could be used to modify disturbance and management practices to increase profitability and diminish environmental impact (Whelan and McBratney, 2000; Bergen et al., 2001). In the Chaco, the prescriptions for restoring a grassy state will be modified by the relative proportion of ecological sites in a paddock, i.e. its landscape features. Success of prescribed fire in increasing treatment longevity will likely occur when there is a large proportion of lowland and midland than upland ecosites. On the other hand, the probability of getting large amounts of herbaceous standing biomass by a successful establishment of Green panic is associated to the area covered by the upland ecosite and final tree density.

Acknowledgements

Funding for this research was provided by the following institutions and projects: Instituto Nacional de Tecnología Agropecuaria, Centro Regional Tucumán-Santiago del Estero, Regional Project '*Intensificación de la Producción de Carne Bovina en el NOA*', 1996–2005, and Universidad Nacional de Santiago del Estero, Secretaría de Ciencia y Técnica, Proyecto Interinstitucional con Transferencia '*Modificación de ecosistemas degradados para uso ganadero y caprino*', 1998–1999. Wade Anderson should also be thanked for his help in English language and an anonymous reviewer for its suggestions that greatly improved the manuscript and the interpretation of results.

References

Adámoli, J., Neumann, R., Colina, A., Morello, J., 1972. El Chaco aluvional salteño. Revista de Investigaciones Agropecuarias Serie 3 9, 165–237.
Aguilera, M., Steinaker, D., 2001. Rolado y siembra en arbustales semiáridos de San Luis. In: Panigatti, J., Buschiazzo, D., Marelli, H. (Eds.), Siembra Directa II. INTA, Buenos Aires, pp. 289–302.
Alessandria, E., Karlin, U., Casermeiro, J., Ferreyra, R., 1987. Repuesta de algunos caracteres estructurales de la vegetación resultante de un rolado ante diferentes presiones de pastoreo. Memoria 1eras Jornadas Nacionales de Zonas Áridas y Semiáridas, pp. 339–341.
Anderson, D., del Aguila, J., Marchi, A., Vera, J.C., Orionte, E., Bernardón, A., 1980.Manejo racional de un campo en la región árida de Los Llanos de La Rioja, INTA, Bs, As.
Anriquez, A., Albanesi, A., Kunst, C., Ledesma, R., López, C., RodriguezTorressi, A., Godoy, J., 2005. Rolado de fachinales y calidad de suelos en el Chaco occidental, Argentina. Ciencia del Suelo (Argentina) 23, 145–157.
Ansley, J., Wiedemann, H., 2008. Reversing the woodland steady state: Vegetation responses during restoration of Juniperus-dominated grasslands with chaining and fire. In: Van Auken, O.W. (Ed.), Western North American Juniperus Communities: A Dynamic Vegetation Type. Springer Verlag, Berlin, pp. 272–290.
Ansley, J., Wiedemann, H., Castellano, M., Slosser, J., 2006. Herbaceous restoration of Juniper dominated grasslands with chaining and fire. Rangeland Ecol. Manage. 59, 171–178.
Ansley, R.J., Rasmussen, G., 2005. Managing native invasive juniper species using fire. Weed Technol. 19, 517–522.
Asner, G., Elmore, A., Olander, L., Martin, R., Thomas Harris, A., 2004. Grazing systems, ecosystem responses and global change. Ann. Rev. Environ. Resour. 29, 261–299.
Bates, J., Miller, R., Svejcar, T., 2000. Understory dynamics in cut and uncut western juniper woodlands. J. Range Manage. 53, 119–126.
Bates, J., Miller, R., Svejcar, T., 2005. Long-term successional trends following western Juniper cutting. Rangeland Ecol. Manage. 58.
Bergen, S., Bolton, S., Fridley, J., 2001. Design principles for ecological engineering. Ecol. Eng. 18, 201–210.
Boetto, M., Alessandría, E., Maccagno, P., Díaz, M., 1999. Control químico de *Acacia furcaptispina*Burkart en un sistema ganadero. Revista Argentina de Producción Animal 19, 411–423.
Boletta, P., 1988. Clima. Cap. 1, In: Desmonte y habilitación de tierras en la Región Chaqueña semiárida. R. Casas, ed. Red de Cooperación Técnica en el uso de los Recursos Naturales en la Región Chaqueña Semiárida, FAO, Santiago, Chile, pp. 7–21.
Boletta, P., Ravelo, A., Planchuelo, A., Grilli, M., 2006. Assessing deforestation in the Argentine Chaco. For. Ecol. Manage. 228, 108–114.
Bond, W., Midgley, J., 2001. Ecology of sprouting in woody plants: the persistence niche. Trends Ecol. Evol. 16, 45–51.
Bozzo, J., Beasom, S., Fulbright, T., 1992. Vegetation response to two brush management practices in South Texas. J. Range Manage. 45, 170–175.
Bray, J., Curtis, J., 1957. An ordination of the upland forest communities of southern Wisconsin. Ecol. Monogr. 27, 325–349.
Britton, C., Wright, H., Dahl, B., Ueckert, D., 1987. Management of tobosa grass rangeland with prescribed fire. Management Note 12. Dept. of Range and Wildlife, TTU, Lubbock, TX, USA.
Bucher, E., Huszar, P., 1999. Sustainable management of the Gran Chaco of South America: ecological promise and economic constraints. J. Environ. Manage. 57, 99–108.
Bucher, E., 1982. Chaco and Caatinga. South American arid savannas, woodlands and thickets. In: Huntley, B., Walker, B. (Eds.), Ecology of Tropical Savannas. Ecol. Studies, vol. 42. Springer Verlag, Berlin, pp. 48–79.
Buhk, C., Retzer, V., Beierkuhnlein, C., Jentsch, A., 2007. Predicting plant species richness and vegetation patterns in cultural landscapes using disturbance parameters. Agric. Ecosyst. Environ. 122, 446–452.
Burkart, A., 1969. Flora ilustrada de Entre Ríos (Argentina): Parte II: Gramíneas. INTA Colección Científica Tomo VI, parte II, Buenos Aires, 551 p.
Burrows, W., 2002. Seeing the wood (land) for the trees—an individual perspective of Queensland woodland studies (1965–2005). Trop. Grasslands 36, 202–217.
Cabral, A., De Miguel, J., Rescai, A., Schmitz, M., Pineda, F., 2003. Shrub encroachment in Argentinean savannas. J. Veg. Sci. 14, 145–152.
Carpenter, S., 1998. The need for large scale experiments to assess and predict the response of ecosystems to perturbation. In: Pace, M., Groffman, P. (Eds.), Success, Limitations and Frontiers in Ecosystem Science. Springer Verlag, NY, pp. 287–312.
Casas, R., Irurtia, C., Michelena, R., 1978. Desmonte y habilitación de tierras para la producción agropecuaria en la República Argentina. INTA-CIRN, 114 p.
Casillo, J., Kunst, C., Ledesma, R., 2006. Control de Celtisspp (tala) mediante fuego prescripto. Ecofuego, IIª Reunión Patagónica y IIIª Reunión Nacional sobre Ecología y Manejo del Fuego, Chubut, Argentina, pp. 185–188.
Conover, W., 1980. Practical Nonparametric Statistics. John Wiley & Sons, New York, 493 pp.
Díaz, M., 2007. Utilización de pastizales naturales. Encuentro Grupo Editor, Córdoba, Argentina, 456 p.
Fensham, R., Dwyer, J., Eyre, T., Fairfax, R., Wang, J., 2011. The effect of clearing on plant composition in mulga. Austral Ecol., doi:10.1111/j.1442-9993.2011.02261.x.
Fumagalli, A., Kunst, C., Pérez, H., 1997. Intensificación de la producción de carne en el NOA. In: Carrozzino, J., Rearte, D. (Eds.), Memoria del 1er. Congreso Nacional de Producción Intensiva de Carne. INTA – ForumArg. de Forrajes, Buenos Aires, pp. 53–61.
Galera, M., 1990. Dinámica y manejo de bosques xerofíticos del chaco árido y comunidades sustitutivas post-tala, rolado, con pastoreo en el Noroeste de la Provincia de Córdoba. Argentina. Memoria Taller interregional Africa/America Latina, MAB, Unesco, Chile.
Gifford, R., Howden, M., 2001. Vegetation thickening in an ecological perspective: significance to national greenhouse gas inventories. Environ. Sci. Policy 4, 59–72.
González, C., Látigo, G., 1981. Rootplowing, front-end stacking, and seeding effects on herbaceous plant species composition. J. Range Manage. 34, 460–465.
Hamilton, W., Ueckert, D., 2004. Rangeland woody plant and weed management-past, present and future. In: Hamilton, W., et al. (Eds.), Brush Management. Past, Present and Future. Texas A&M University Press, College Station, TX, pp. 3–13.
Heitschmidt, R., Pieper, R., 1983. Integrated brush control and grazing management. In: Mc Daniel, K. (Ed.), Proceedings Brush Management Symposium Society for Range Management. Albuquerque, New Mexico, pp. 89–95.
Henkin, Z., Noy-Meir, I., Kafkafi, U., Gutman, M., 1998. Rehabilitation of Mediterranean dwarf-shrub range-land with herbicides, fertilizers, and fire. J. Range Manage. 51, 193–199.
Holl, K.D., 2010. Writing for an international audience. Restor. Ecol. 18, 135–137.
Hurlbert, S., 2004. On misinterpretation of pseudoreplication and related matters: a reply to Oksanen. Oikos 104, 591–597.
Instituto de Botánica Darwinion, 2011. Catálogo de Plantas Vasculares de la República Argentina. <www.darwin.edu.ar/Proyectos/FloraArgentina>.
Johnson, D., 2006. The many faces of replication. Crop Sci. 46, 2486–2491.
Johnson, E., Miyanishi, K., 2008. Testing the assumptions of chronosequences in succession. Ecol. Lett. 11, 419–431.
Krebs, C., 1999. Ecological Methodology. Addison Wesley Longman, Inc, xii + 620 pp.
Kunst, C., Ledesma, R., Basan, M., Angella, G., Prieto, D., Godoy, J., 2003. Rolado de fachinales e infiltración de agua en el suelo en el Chaco occidental argentino. Revista de Investigaciones Agropecuarias 32, 105–122.
Kunst, C., Monti, E., Perez, H., Godoy, J., 2006. Assessment of rangelands of southwestern Santiago del Estero for management and research. J. Environ. Manage. 80, 248–265.
Kunst, C., Bravo, S., Moscovich, F., Herrera, J., Godoy, J., Vélez, S., 2001. Control de tusca (*Acacia aroma* Gill ap. H. et A.) mediante fuego prescripto. Revista Argentina de Producción Animal 20, 199–213.
Le Brocque, A., Goodhew, K., Zammit, C., 2009. Overstorey tree density and understorey regrowth effects on plant composition, stand structure and floristic richness in grazed temperate woodlands in eastern Australia. Agric. Ecosyst. Environ. 129, 17–27.
Ledesma, R., 2006. Desarrollo de sistemas ganaderos: una alternativa de manejo en ecosistemas degradados del Chaco semiárido, Universidad Nacional de Santiago del Estero, Santiago del Estero, Argentina. Masters Thesis, 134 p.
Littell, R.C., Henry, P.R., Ammerman, C.B., 1998. J. Anim. Sci. 76, 1216–1231.
Mc Cune, B., Grace, J., 2002. Analysis of Ecological Communities. MJM Software Design, OR, USA.
McDonald, J.H., 2009. Handbook of Biological Statistics, 2nd ed. Sparky House Publishing, Baltimore, MD, Consulted 12/12/2010 in http://udel.edu/~mcdonald/statnested.html.
McIver, J., Starr, L., 2001. Restoration of degraded lands in the interior ColumbiaRiver basin: passive vs. active approaches. For. Ecol. Manage. 153, 15–28.
McIvor, J., 2006. Pasture management in semiarid tropical woodlands: effects on tree regrowth. Trop. Grasslands 40, 14–23.
Menges, E., Gordon, D., 2010. Should mechanical treatments and herbicides be used as fire surrogates to manage Florida uplands? A review. Florida Sci. 73, 147–174.
Mitchell, J., 1983. Analysis of forage production for assessments and appraisals. U.S.D.A. – Forest Service GTR RM-98.
Moore, M., Casey, C., Bakker, J., Springer, J., Fulé, P., Covington, W., Laughlin, D., 2006. Herbaceous vegetation responses (1992–2004) to restoration preatments in a Ponderosa pine forest. Rangeland Ecol. Manage. 59, 135–144.
Morello, J., Saravia Toledo, C., 1959. El bosque chaqueño I y II. Rev. Agronómica del Noroeste Argentino, 3, 5–81/209–258.

Moustakas, A., Sakkos, K., Wiegand, K., Ward, D., Meyer, K., Eisinger, D., 2009. Are savannas patch-dynamic systems? A landscape model. Ecol. Model. 220, 3576–3588.
Nai Bregaglio, M., Karlin, U., Coirini, R., 1999. Repuesta de especies arbustivas forrajeras a tratamientos de desmonte selectivo en el Chaco árido, Argentina. Multequina 8, 101–109.
Navall, M., 2008. Rolados y manejo forestal. In: Kunst, C., Ledesma, R., Navall, M. (Eds.), RBI-Rolado selectivo de baja intensidad. INTA EEA Santiago del Estero, Informe Técnico No 57. Santiago del Estero, pp. 72–85.
Naveh, Z., 1990. Fire in the Mediterranean—a landscape ecological perspective. In: Goldammer, J., Jenkins, M. (Eds.), Fire in Ecosystem Dynamics. SPB Academic Publishing, The Hague, Netherlands, pp. 1–20.
Naveh, Z., 2004. Multifunctional, self organizing biosphere landscapes and the future of our total human ecosystem. World Futures 60, 469–503.
Noble, J., Muller, W., MacLeod, N., Boduluvic, Z., Jones, P., Wood, J., 2005. Integrated shrub management in semi-arid woodlands of eastern Australia: ground and aerial application of defoliant to shrubs regenerating after disturbance. Rangeland J. 27, 117–134.
Nolte, K., Gabor, T., Hehman, M., Asleson, M., Fulbright, T., Rutledge, J., 1994. Long term effects of brush management on vegetation diversity in ephemeral drainages. J. Range Manage. 47, 457–459.
Norkko, A., Rosenberg, R., Thrush, S., Whitlatch, R., 2006. Scale and intensity-dependent disturbance determines the magnitude of opportunistic response. J. Exp. Mar. Biol. Ecol. 330, 195–207.
Passera, C., Borsetto, O., Allegreti, L., 1996. Short term effects of shrub control on two different plant communities in Argentina. J. Arid Environ. 34, 415–420.
Pastorok, R., MacDonald, A., Sampson, J., Wilber, P., Yozzo, D., Titre, J., 1997. An ecological decision framework for environmental restoration projects. Ecol. Eng. 9, 89–107.
Platt, W., Connell, J., 2003. Natural disturbances and directional replacement of species. Ecol. Monogr. 73, 507–522.
Pretzsch, H., 2007. Analyzing and modeling forest stand dynamics for practical application: a European review and perspective. Eurasian J. For. Res. 10, 1–17.
Ragonese, A., 1951. La vegetación de las Salinas Grandes. Revista de Investigaciones Agrícolas 5. INTA, Buenos Aires, pp. 3–232.
Renolfi, R., Fabiani, A., Kunst, C., Pérez, H., Dalla Tea, F., 1987. Estimación de la altura de uso adecuado de gramíneas nativas de la región chaqueña occidental (Sgo del Estero). Revista Argentina de Producción Animal 7, 181–189.
Rodríguez Rey, J., Rovati, A., Guzmán, L., Juárez V. 1983. Recuperación de pasturas naturales de monte mediante el uso de herbicidas hormonales y clausura. Pub. Misc. 75. Est. Agro-Industrial Obispo Colombres, Tucumán, Argentina.
Rollins, D., Bryant, F., 1986. Floral changes following mechanical brush removal in Central Texas. J. Range Manage. 39, 237–240.
Ryan, K., Noste, N., 1985. Evaluating prescribed fires. In: Lotan, J., Kilgore, B., Fischer, W., Mutch, R. (Eds.), Proceedings of the Symposium and Workshop on Wilderness Fire. Intermountain Forest and Range Exp. Station, Missoula, Montana, GTR INT-182, pp. 230–238.
SAS, 2002. Release 6.20 Edition. SAS Institute Inc., Cary, NC.
Sawadogo, L., Tiveau, D., Nygard, R., 2005. Influence of selective tree cutting, livestock and herbaceous biomass in the savannah woodlands of Burkina Faso, West Africa. Agric. Ecosyst. Environ. 105, 335–345.
Schindler, D., 1998. Replication versus realism: the need for ecosystem-scale experiments. Ecosystems 1, 323–334.
Smit, G., 2004. An approach to tree thinning to structure southern African savannas for long term restoration from bush encroachment. J. Environ. Manage. 71, 179–191.
Smit, G., 2005. Tree thinning as an option to increase herbaceous yield of an encroached semiarid savanna in South Africa. BMC Ecol. 5, 1–15, doi:10.1186/1472-6785-5-4.
Sousa., W., 1984. The role of disturbance in natural communities. Ann. Rev. Ecol. Syst. 15, 353–391.
TannerF G., Wood, J., Kalmbacher, R., Martin, F., 1988. Mechanical shrub control on flatwoods range in south Florida. J. Range Manage. 41, 245–248.
Teague., W., Grant, U., Kreuter, H., Diaz-Solis, S., Dube, M., Kothmann, M., Pinchak, W., Ansley, R., 2008. An ecological economic simulation model for assessing fire and grazing management effects on mesquite rangelands in Texas. Ecol. Econ. 64, 611–624.
Thom, R., 1997. System development matrix for adaptive management of coastal ecosystem restoration projects. Ecol. Eng. 8, 219–232.
Tortorelli, L., 2009. Maderas y bosques argentinos, 2ª edición. Orientación Grafica Editora, 592 p.
Tothill, J., Hargreaves, J., Jones, R., 1978. *BOTANAL*. A comprehensive sampling and computing procedure for estimating pasture yield and composition. I. Field sampling. CSIRO, Australian Div. Trop. Crops and Pastures. Brisbane, Queensland, Trop. Agron. Tech Mem. 8.
UCLA (University of California, Los Angeles, Academic Technology Services, Statistical Consulting Group), 2010. Introduction to SAS. 'How can I compare regression coefficients across three (or more) groups?' http://www.ats.ucla.edu/stat/sas/faq/compreg3.htm. Consulted 5/19/2010.
Underwood, A., 1998. Experiments in Ecology. Cambridge University Press, Cambridge, United Kingdom, xviii + 504 pp.
University of England, School of Psychology, Web Stat, 2011. Chapter 7: Analysing the data, Part IV, Analysis of variance: Two-way Analysis of variance, Two-way analysis of variance. http://www.une.edu.au/WebStat/unit_materials/c7_analysis of variance/twoway_analysis of variance.htm, Consulted 9/15/2011.
Van Auken, O., 2009. Causes and consequences of woody plant encroachment into western North American grasslands. J. Environ. Manage. 90, 2931–2942.
Watts, A., Tanner, G., Dye, R., 2006. Restoration of dry-prairie using fire and roller chopping. Land of fire and water: the Florida dry prairie ecosystem. In: Noss, R. (Ed.), Proceedings of the Florida Dry Prairie Conference., pp. 225–230.
Welch, T., 2000. Brush management methods. Agri-Life Extension, Texas A&M System. Consulted in http://repository.tamu.edu/bitstream/handle/1969.1/86801/pdf_106.pdf?sequence=1.
Whelan, B., McBratney, A., 2000. The 'null hypothesis' of precision agriculture management. Precision Agriculture 2, 265–279.
White, P., Jentsch, A., 2001. The search for generality in studies of disturbance and ecosystem dynamics. Prog. Bot. 62, 399–450.
Willcox, E., Giuliano, W., 2010. Seasonal effects of prescribed burning and roller chopping on saw palmetto in flatwoods. For. Ecol. Manage. 259, 1580–1585.
Williams, D., Baruch, Z., 2000. African grass invasion in the Americas: ecosystem consequences and the role of ecophysiology. Biol. Invas. 2, 123–140.
Wolters, G., 1981. Timber thinning and prescribed burning as methods to increase herbage on grazed and protected longleaf pine ranges. J. Range Manage. 34, 494–497.
Zedler, J., 2007. Success: an unclear, subjective descriptor of restoration outcomes. Ecol. Restor. 25, 162–168.