

Effects of forest preservation, livestock exclusion and use of shrubs as potential nurses on planting success of an endangered tree

Forest preservation, livestock and planting success

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#### Author contributions

DR, RT conceived the research idea; DR collected seeds and produced the saplings; DR, JP, TV, LH, SN, IIG set up the field experiment; RT performed statistical analyses; RT wrote the paper with contributions from DR, TV, LH, SN; all authors collected data, discussed the results and made comments on the manuscript.

#### Abstract

Domestic livestock are widespread in seasonally dry forests, likely causing forest degradation and limiting tree seedling establishment. Shrubs can play an important role in facilitating tree regeneration, by protecting trees from livestock damage and ameliorating unfavorable abiotic conditions. We aimed at disentangling the relative contribution of grazing exclusion, long-term forest conservation, and the potential facilitation effect of shrubs on the performance of saplings of the native tree *Kageneckia lanceolata*. We planted 400 saplings in grazed and ungrazed areas situated both in a preserved and a degraded forest. In each situation, we established planting plots in three accompanying vegetation treatments: herbs, a non-leguminous spiny shrub and a leguminous spiny shrub. Survival of three-year-old saplings was 10-fold higher in the preserved than in the degraded forest and two-fold higher in the ungrazed than in the grazed site. Differences in survival among accompanying vegetation treatments were much smaller than between

grazing treatments. Survival significantly increased with increasing protection by shrubs only in the degraded site. Sapling growth patterns were fairly similar to survival patterns, with no growth in the degraded forest, except for limited growth under both shrubs in the ungrazed site. We conclude that, in selecting plantation sites for the study species, forest condition and grazing exclusion should be prioritized over microsite selection based on neighboring vegetation.

Keywords: grazing, soil nutrients, facilitation, spiny shrub, nitrogen-fixing shrub, forest degradation.

### **Implications for practice**

To increase population size of an endangered tree species, selecting planting sites in preserved forests with little or no livestock grazing may be a much better strategy than selecting microsites based on neighboring vegetation.

Selecting planting sites using microsite selection should be useful to enrich degraded forests; however, a low success rate should be expected and grazing must be completely excluded.

### **Introduction**

Free-ranging domestic livestock are widespread around great portions of the globe, with both negative and positive effects on forest regeneration (Mazzini et al. 2018). Negative effects include trampling of saplings, bud consumption, and changes in the characteristics of microsites, all of which can reduce sapling establishment (Griscom et al. 2005; Goheen et al. 2007; Etchebarne & Brazeiro 2016). In turn, positive effects include seed dispersal and reduction of competition from grasses and herbs (Goheen et al. 2010; Mazzini et al. 2018).

Long-term livestock grazing is one of the main causes of vegetation and soil degradation in many areas of the globe, creating unfavorable abiotic conditions for tree establishment (Säumel et al. 2011; Schulz et al. 2018). Livestock cause soil compaction as well as soil erosion in regions where grazing is combined with prescribed fires used to stimulate plant regrowth (Castellano & Vallone 2007; Oesterheld et al. 2011; Cingolani et al. 2013). In seasonally dry mountain forests, the high soil exposure at the end of the dry season, intense early rains and steep slopes that favor runoff contribute to soil erosion. In turn, soil erosion reduces soil nutrients and further increases water runoff, reducing water availability to plants (Cingolani et al. 2013; Ellison et al. 2017; Sparacino et al. 2020). As a consequence, seasonally dry forests grazed for long periods are composed of a mosaic of physically degraded environments induced by grazing-related erosion combined with relatively fertile forest islands with denser woody vegetation and lower livestock densities (Howard et al. 2012; Bernardi et al. 2019).

In this context, remnant woody vegetation can play an important role in facilitating tree establishment, ameliorating both the immediate negative herbivore

effects, such as browsing and trampling (Padilla & Pugnaire 2006; Cuevas et al. 2013), and improving degraded soil conditions due to long-term livestock grazing (Morgan 2005; Howard et al. 2012). Previous studies in seasonally dry forests suggest facilitative interactions through protection from herbivores, as evidenced by a spatial aggregation between spiny shrubs and tree seedlings (Táalamo et al. 2015) and by an increase in survival and growth of planted tree saplings under shrubs (Torres & Renison 2015; 2016). Furthermore, shrubs can facilitate tree establishment by mitigating abiotic conditions, such as soil erosion and water runoff (Flores & Jurado 2003; Torres et al. 2008; Gómez-Aparicio 2009). Nitrogen-fixing plants might be important for the establishment of other species in degraded areas, since they can provide N-rich leaf litter (Padilla & Pugnaire 2009). Previous reviews showed that N-fixing woody species increase soil N and understory biomass in arid environments (Blaser et al. 2013), and that N-fixing plants occur much more frequently as nurses than as beneficiaries (Bonanomi et al. 2011).

Planting success, estimated as survival of planted material, is highly variable in seasonally dry mountain forests species, with records of first year survival ranging between 0.2 and 0.9 (Natale et al. 2014; Torres & Renison 2017). Drought-induced dieback results in negative growth, i.e. a reduction in sapling height from planting date to monitoring date (Gerhardt 1996; Verzino et al. 2004; Torres & Renison 2015; 2016). Hence, there is a need for research about facilitation by neighboring plants to improve planting success, mainly in seasonally dry mountain forests used by free-ranging domestic livestock. Therefore, we asked the following questions: (1) which is the most important factor limiting the establishment of an endangered tree: livestock browsing of

saplings or nutrient-depleted soil due to long-term forest degradation? (2) Can we overcome those limitations by using shrub species as nurse plants?

Here, we assessed the success of planting a native tree species under three accompanying vegetation treatments, in grazed and ungrazed sites, in each of two forests differing in grazing history and herbivore density (a preserved and a degraded forest). We selected the target species *Kageneckia lanceolata*, an endangered tree native to Chaco Serrano region in South America; as accompanying vegetation we selected existing herbaceous vegetation and two native spiny woody species widely distributed in degraded forests: the leguminous *Vachellia caven* and the non-leguminous *Condalia montana*. Our aims were to (1) compare the performance (in terms of survival and growth) of *Kageneckia lanceolata* saplings planted in grazed and ungrazed sites both in a preserved and a degraded forest; (2) determine the potential facilitation effect of two native shrub species used as nurse plants on the performance of *Kageneckia lanceolata* saplings planted in the grazed and ungrazed sites both in the preserved and in the degraded forests; (3) determine browsing damage to saplings and soil chemical characteristics under the different accompanying vegetation and grazing treatments both in the degraded and preserved forests.

We hypothesize that livestock grazing would have negative effects on *Kageneckia lanceolata* sapling performance through sapling browsing; in areas degraded by long-term grazing (hereafter, degraded forest), sapling performance would also be negatively affected by poor soil nutrient conditions. Spiny shrubs could attenuate short-term negative effects of livestock by protecting saplings from browsing, whereas spiny shrubs

that are also N-fixing could both protect saplings from browsing and attenuate poor nutrient (mainly N) conditions. Hence, we predict that the performance of our selected target species, estimated as the survival and growth of planted saplings: (1) will be higher under ungrazed than under grazed conditions; (2) will be higher in a preserved forest than in a degraded forest; (3) will differ among accompanying vegetation treatments according to degradation and grazing conditions, with N-fixing shrubs being beneficial only in the degraded forest, and spiny shrubs, only under grazed conditions; (4) saplings will be browsed only in grazed sites, but to a lesser degree under both spiny shrubs and leguminous spiny shrubs than under herbs; and (5) soil nutrient content will be higher under shrubs, particularly under leguminous spiny shrubs, than in herbaceous vegetation.

## Methods

### Study region

The vegetation in the mountains of central Argentina corresponds to the Chaco Serrano district (Oyarzabal et al. 2018) of the Gran Chaco Region. Chaco Serrano is an open canopy forest in exposed and disturbed sites, and a dense canopy forest in less disturbed sites; the latter is often found within ravines, where fires and livestock are less frequent. Mature stands at altitudes from 500 to 1800 m asl are dominated by the trees *Schinopsis lorentzii* (Anacardiaceae) and *Lithraea molleoides* (Anacardiaceae), which coexist with different tree species, such as *Zanthoxylum coco* (Rutaceae), *Vachellia caven* (Fabaceae),

*Geoffroea decorticans* (Fabaceae), *Condalia montana* (Rhamnaceae), *Schinus fasciculatus* (Anacardiaceae), *Celtis ehrenbergiana* (Cannabaceae), and *Kageneckia lanceolata* (Rosaceae) (Giorgis et al. 2011a; Cabido et al. 2018). Chaco Serrano forests are being rapidly degraded by recurrent wildfires, extensive livestock grazing, housing developments, non-native tree plantations, and multiple foci of invasive non-native species (Hoyos et al. 2010; Giorgis et al. 2011b; Argañaraz et al. 2015). Preserved native forests in the mountains of central Argentina have been reduced to less than 12% of the original area (Zak et al. 2004), and soils are being lost at very high rates in the most susceptible areas at higher altitudes (Cingolani et al. 2013).

The climate is subtropical, with mean temperatures of 16.6 °C at 700 m asl and mean annual precipitations of 771 mm for the 1995 and 2017 period, with wide annual variations ranging from 474 to 1201 mm; precipitations are concentrated in the warm season, from September to March. Precipitations during our study period (2015, 2016 and 2017) were 1021, 873 and 727 mm, respectively, i.e. above the average for the former two years and slightly below for the latter (R. Renison and A. M. Cingolani, unpublished data; Abril & Zanvettor 2012).

We selected two experimental areas: (1) a relatively preserved forest in “Reserva Natural Vaquerías” (S -31.12° W -64.44°, 1000 m asl, 27° slope inclination, SW slope aspect) and (2) a relatively degraded forest in “San Antonio” (S -31.48° W -64.54°, 700 m asl, 14° slope inclination, SE slope aspect) (Figure 1a). The relatively preserved forest (hereafter, preserved forest) has been managed as a conservation area by the National University of Córdoba since 1970. The vegetation is an open woodland co-dominated by *Lithraea molleoides*, *Zanthoxylum coco*, *Schinus fasciculatus*, *Vachellia caven*, *Condalia*



*montana*, and with the presence of *Kageneckia lanceolata* at very low densities. Mean height of the tree canopy is about 7-9 m. Non-native invasive tree species, such as *Gleditsia triacanthos* (Fabaceae), *Pyracantha angustifolia* (Rosaceae) and *Ligustrum lucidum* (Oleaceae), cover 30% of the surface area, with the highest densities occurring in ravines and along streams (Salazar et al. 2013). The preserved forest had domestic livestock, mainly horses but also cattle and goats, at low densities of approximately 0.05 cattle equivalents.ha<sup>-1</sup> (personal estimations; unpublished data); other wild herbivores present at unknown densities include the native deer (*Mazama gouazoubira*), the collared peccary (*Pecari tajacu*) and the introduced European hares (*Lepus europaeus*), as reported in the reserve management plan (Kufner et al. 2012) and by the local warden (J. Piedrabuena 2015, park ranger, Natural Reserve Vaquerías, National University of Córdoba, personal communication).

In the relatively degraded forest (hereafter, degraded forest), the vegetation consisted of grasslands with patches of woody species, such as *Lithraea molleoides*, *Vachellia caven*, *Geoffroea decorticans*, *Condalia montana*, *Schinus fasciculatus*, *Celtis ehrenbergiana*, and several non-native species, mainly *Ulmus pumila* (Ulmaceae), *Gleditsia triacanthos* (Fabaceae) and *Pyracantha angustifolia* (Rosaceae). Here, we found only one *Kageneckia lanceolata* tree. The degraded forest also presented sites with bare soil and short grasses resulting from domestic livestock grazing. Mean height of the tree canopy was about 3-4 m. This experimental area has been affected by several fires and livestock presence since about the 1950s. During our study period, livestock consisted mainly of cattle with a few horses at a density of 0.5 cattle equivalents.ha<sup>-1</sup>, as estimated from our direct counts. We detected no signs of wild native herbivores, such as

wild deer or peccaries, and observed the introduced European hare only occasionally (personal observations).

Our target study species, *Kageneckia lanceolata*, hereafter *Kageneckia* (Rosaceae), is an evergreen tree of up to 5 m in height, distributed in Peru, Bolivia and Argentina from latitudes of -7 to -33 degrees and a latitudinal extent of about 3200 km. The fruit is a pentamerous star-shaped capsule, about 2-3 cm in diameter, and the seeds are winged and dispersed by wind (Demaio et al. 2015). The species has been listed as Vulnerable by the World Conservation Monitoring Center (<http://www.iucnredlist.org/details/39030/0>, downloaded on 26 April 2020), with the main threats being grazing, clearing and fuel wood extraction.

#### Experimental setup

In February 2015, in each experimental forest site (preserved and degraded), we established two adjacent plots: a fenced livestock enclosure (ungrazed site) and an adjacent unfenced site where livestock grazed freely (grazed site) (Figure 1b). Within each grazed and ungrazed plot, we randomly established 100 50 x 50-cm planting plots distributed in three accompanying vegetation treatments: (1) herbaceous vegetation (no shrub) of less than 1 m in height (40 plots), (2) under a non-leguminous spiny shrub (*Condalia montana*) of up to 3.5 m in height (30 plots), and (3) under a leguminous spiny shrub (*Vachellia caven*) of up to 6.5 m in height (30 plots) (Figure 1c). We planted two saplings of *Kageneckia* per plot to ensure sufficient surviving saplings for growth analysis. We established a higher number of plots in herbaceous vegetation, based on the

high mortality reported for other species (e.g. Torres & Renison 2016). Distance between planting plots varied between 3 and 5 m. We irrigated each plot only once, at outplanting, with 5 L of water. At outplanting, *Kageneckia* saplings were 19 months old,  $44.6 \pm 0.8$  cm in mean height, and had been acclimatized to the harsher conditions outside the nursery for 30 days. Details on the production of these *Kageneckia* saplings are provided in Renison et al. (2019).

We monitored survival and height, as indicators of sapling performance, 2, 9, 14, 33 and 38 months after planting. To better interpret our results, we also monitored the presence/absence (1/0) of browsed buds, and estimated soil nutrient content in three composite soil samples per combination of treatments (3 accompanying vegetation treatments x 2 grazing treatments x 2 forest conditions x 3 replicates = 36 soil samples). Each composite soil sample was composed of 10 thoroughly mixed plot subsamples taken from a depth of up to 10 cm in February 2018. Soil samples were maintained in a refrigerator at 4 °C until analysis. For each sample, we determined (1) nitrates (ppm) with phenol-disulphonic method Bolotina & Abramova (1968), (2) organic matter (%) with the Walkley & Black (1934) method, and (3) extractable phosphorus (ppm) with Bray & Kurtz I method (Jackson 1964).

#### Statistical analysis

All statistical analyses were performed using data from the last monitoring date, i.e. 38 months after planting. We ran Generalized Linear Models (GLMs) for most response variables. We assumed a binomial distribution for sapling survival and browsing damage

(number of times a browsed sapling was recorded.total records<sup>-1</sup>) and a normal distribution for sapling growth (difference between final and initial height), soil nitrate and soil organic matter. Six plots where tagging was lost were removed from analysis. Since we planted two saplings per plot, for survival analysis we considered the survival of one randomly chosen sapling per plot; for sapling growth and browsing damage, we followed the same procedure when two surviving saplings per plot were available on the last monitoring date. We included the accompanying vegetation, grazing and forest condition as fixed factors, as well as all the interactions among factors. We ran Fisher's LSD and DGC post-hoc tests to compare treatments. We used the Kruskal-Wallis test to analyze differences in soil extractable phosphorus among treatments. All analyses were performed using Infostat 2018 statistical package; and in all cases, significance level was 0.05 (Di Rienzo et al. 2015).

## Results

Planting success of *Kageneckia* saplings showed wide variations. Survival of three-year-old saplings was 10-fold higher in the preserved forest than in the degraded forest (mean for both grazing situations: 62.0 and 6.3 %, respectively, GLM, n = 394,  $\chi^2 = 271.7$ , p < 0.001) and two-fold higher in ungrazed than in grazed sites (mean for both forests; 45.5 and 22.8 %, respectively (p < 0.001). Sapling survival was nearly 25% lower in plots with herbaceous vegetation and under the non-leguminous spiny shrub than under the leguminous spiny shrub (p = 0.01), with differences tending to be smaller in the

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preserved forest than in the degraded forest (accompanying vegetation x forest condition interaction,  $p = 0.06$ ) (Figure 2a).

Similarly to survival, growth of three-year-old saplings was 77 cm higher in the preserved forest than in the degraded forest (66 and -11 cm, respectively; GLM,  $n = 178$ ;  $\chi^2 = 130.8$ ,  $p < 0.001$ ) and was 34 cm higher in ungrazed than in grazed sites (49 and 15 cm, respectively;  $p < 0.001$ ). Growth was negative or null in the degraded forest, except under both shrub species in the ungrazed site. Unlike survival, growth was influenced by the accompanying vegetation x grazing interaction ( $p = 0.003$ ). Indeed, growth tended to decrease from herbaceous vegetation to leguminous spiny shrub in the preserved forest under grazing exclusion, whereas an opposite pattern was detected in the preserved forest with grazing and in the degraded forest under grazing exclusion (Figure 2b). The accompanying vegetation treatment per se was not significant ( $p = 0.15$ ).

In grazed sites, 96% and 100% of saplings were found to be browsed in the preserved and degraded forest, respectively, on at least one monitoring date. In ungrazed sites, 3% and 0% of saplings were browsed in the preserved and degraded forest, respectively, probably by introduced hares or native deer (grazing treatment  $p < 0.001$ , GLM,  $n = 178$ ;  $\chi^2 = 496.7$ ; Figure 2c). Saplings planted in the herbaceous treatment tended to be more heavily browsed than those under both leguminous and non-leguminous spiny shrubs ( $p = 0.09$ ).

Soil nitrate and soil organic matter contents were higher in the preserved than in the degraded forest (GLM,  $n = 36$ , nitrate:  $\chi^2 = 35.4$ ,  $p < 0.001$ ; organic matter:  $\chi^2 = 53.0$ ,  $p < 0.001$ ) and differed among accompanying vegetation treatments (nitrate:  $p < 0.001$ ;

organic matter:  $p = 0.001$ ). Soil nitrate and soil organic matter contents were similar among accompanying vegetation treatments in the preserved forest, whereas in the degraded forest the lowest values were recorded in herbaceous vegetation and the highest values were detected under both leguminous and non-leguminous spiny shrubs (nitrate: forest condition  $\times$  accompanying vegetation interaction,  $p = 0.008$ ; organic matter: forest condition  $\times$  accompanying vegetation interaction,  $p < 0.001$ ; Figures 2d and e). We did not find differences in soil extractable phosphorus among accompanying vegetation treatments, between grazing treatments, or between preserved and degraded forests (min:  $5.53 \pm 0.65$  ppm, max:  $10.41 \pm 3.79$ ; Kruskal Wallis test,  $n = 36$ ,  $H = 9.8$ ,  $p = 0.5$ ).

## Discussion

Our results confirm our hypothesis about the important negative effects of livestock on sapling performance (both survival and growth) through direct browsing and, presumably, even more important negative effects through soil nutrient (nitrate and organic matter) content reduction. Regarding the role of potential nurse shrub species, our results did not support our expectation that sapling performance would differ among accompanying vegetation treatments according to degradation and grazing conditions.

Previous studies carried out in central Argentina also reported negative effects of domestic livestock on survival and growth of native trees (Renison et al. 2015; Tálamo et al. 2015). However, unlike other studies (Gómez-Aparicio 2009; Torres & Renison 2015;

2016), we did not find a consistent decrease in the negative effect of livestock under the protection of other woody plants, possibly because livestock browsing was not reduced effectively under the spiny shrubs used in our experiments. Our results also agree with reports of negative effects of livestock grazing on tree regeneration in other seasonally dry forests (Tsegaye et al. 2009; Wassie et al. 2009).

Planting success was also limited in the degraded forest. Long-term effects of livestock grazing include changes in vegetation cover and composition, as well as in soil microbial community, nutrient content, and moisture (Wassie et al. 2009; Noumi et al. 2015; Zhang et al. 2018a). In this context, the preserved forest can provide protection from solar radiation and frost, as well as increased water content and litter accumulation, with the consequent increase in nutrients (Abiyu et al. 2017). In particular, leguminous plants often provide litter with a high content of nutrients, mainly nitrogen, and facilitate the establishment and growth of other species (Avendaño-Yáñez et al. 2018; Zhang et al. 2018b). Our results partially support these findings, since the preserved forest had higher nitrate and organic matter values than the degraded forest, whereas in the latter, nitrate and organic matter slightly increased from plots with herbaceous vegetation to microsites under shrubs, with the highest values being recorded under the leguminous spiny shrub. However, there was a slight positive effect of shrubs on sapling performance; this result agrees with studies showing that positive interactions among plants are less important under high abiotic stress than under intermediate stress (Malkinson & Tielbörger 2010; Koyama & Tsuyuzaki 2013; Verwijmeren et al. 2014).

Regarding our N-fixing shrub, soil nutrient results support previous findings reporting that *Vachellia caven* has higher nodule efficiency than other native leguminous

species (Aronson et al. 2002). This phenomenon seems to be related to a greater capacity to nodulate with a wide range of strains (Frioni et al. 1998). *Vachellia caven* also improves soil physical properties and has been recommended for protecting sites for sclerophyll forest restoration purposes (Soto et al. 2015). Root-Bernstein et al. (2017) found that *Vachellia caven* plays an important role in the dynamic successional pathways of central Chile, acting as a nurse of endemic sclerophyll tree species.

In conclusion, when selecting plantation sites for the study species, forest condition and grazing exclusion should be considered over microsite selection based on neighboring vegetation. Thus, programs aiming to rescue *Kageneckia lanceolata* from further declining by planting saplings must select preserved forests as plantation sites. Further improvements in planting success in preserved forests may be achieved with little or no livestock grazing. Programs aiming at the restoration of degraded forests using *Kageneckia lanceolata* for enrichment planting must acknowledge planting success will be very poor, and must focus on planting with no livestock grazing and under N-fixing shrubs. We suggest further research testing soil amendments, watering treatments or beneficial microorganisms to improve *Kageneckia* planting success in degraded forests, as suggested by Navarro Ramos et al. (2018). Understanding how disturbances, such as livestock grazing and forest degradation, affect the interaction between woody species may help to design successful nurse-based plantations, using this or other potential nurse species.

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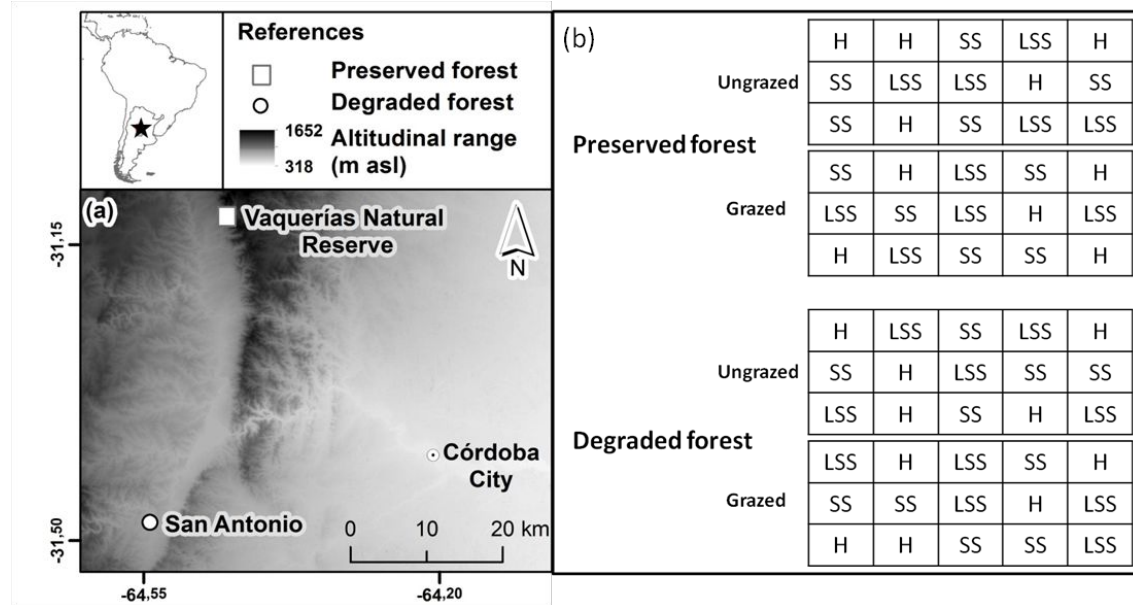


Figure 1. (a) Elevation map of the study region in the mountains of central Argentina. The preserved and degraded forests are indicated with different symbols. Upper left: study region in Argentina, South America (star). (b) Diagram showing the experimental design. Ungrazed plots consisted of livestock enclosures (2 ha each) built in the degraded forest in October 2007 and in the preserved forest in September 2012. Grazed plots consisted of an area adjacent to the enclosure. Small rectangles inside the grazed and ungrazed plots indicate randomly assigned planting plots to three accompanying vegetation treatments: H: plots with herbaceous vegetation, SS: non-leguminous spiny shrub, LSS: leguminous spiny shrub.

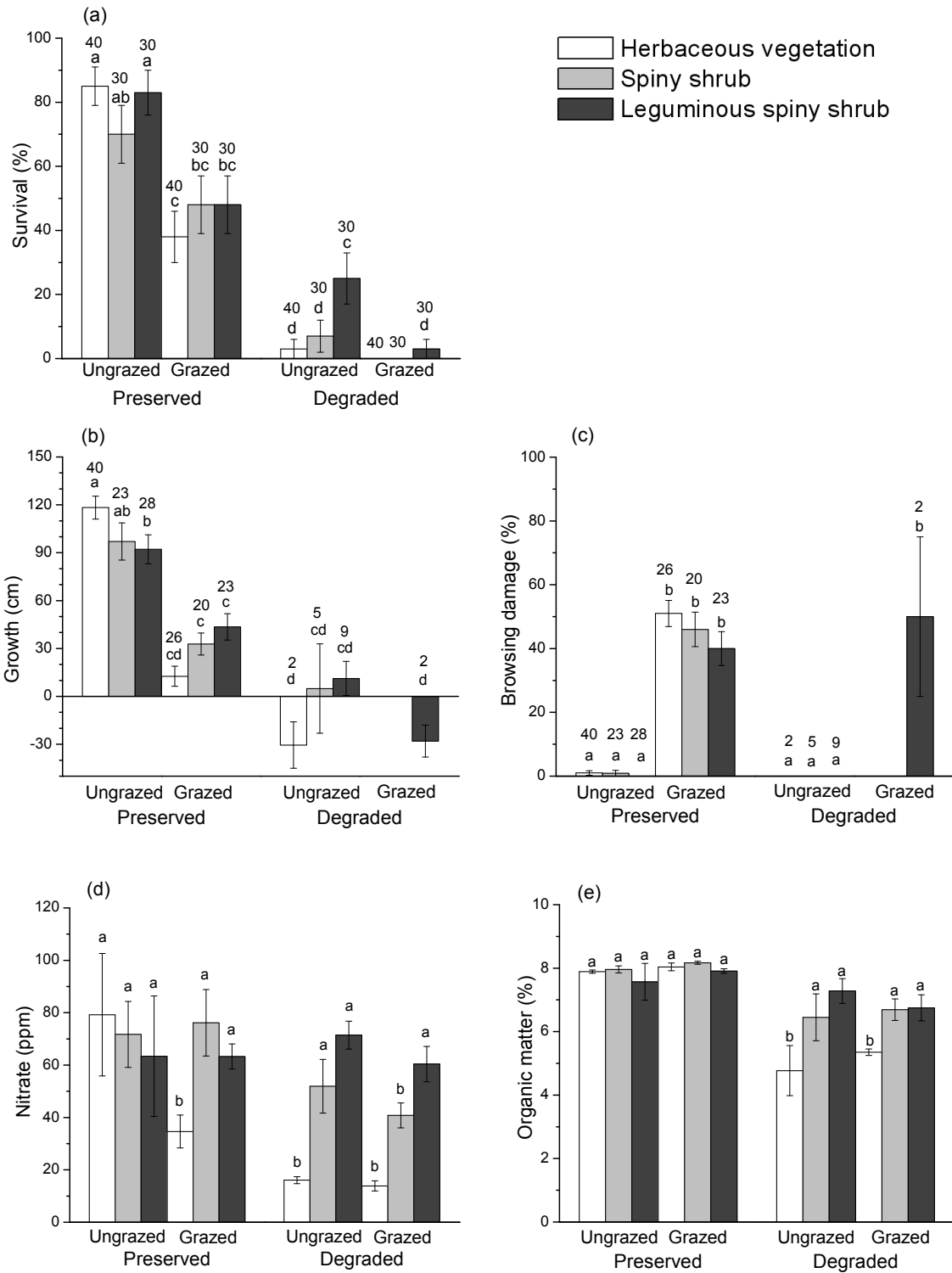


Figure 2. *Kageneckia* sapling survival (a), growth (b), browsing damage (c), soil nitrate content (d), and soil organic matter (e) three years after planting in a preserved and a degraded forest, two grazing treatments and under three accompanying vegetation treatments (see references above right). Bars show mean and  $\pm$ SE. Numbers above bars in figures (a), (b) and (c) indicate the number of saplings per treatment; for figures (d) and (e), the number of replicates was three composite soil samples. Different letters above bars show significant differences (LSD Fisher and DGC post-hoc test).