Ecological Processes

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Retention and redistribution of biological legacies generate resource sinks in silvopastoral systems of Arid Chaco forests

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Abstract

Background: Forests are used for multiple purposes worldwide, which often include timber harvest, firewood extraction and livestock raising. An excessive pressure on multipurpose systems may decrease soil cover, promoting soil erosion and causing the loss of other resources, as litter and seeds. Retention forestry practices can help to decrease or mitigate resource loss in the managed stands. Specifically, retaining and redistributing biological legacies (e.g. logs, branches, woody debris) at strategic locations can create sediment, litter, and seed-sinks in the silvopastoral systems. In addition, grazing management could increase or, even, decrease the success of this practice. In this study, we assessed the effect of branch barriers and grazing management on resource run-off/run-on processes in silvopastoral systems of Arid Chaco (Córdoba, Argentina). To do this, a 2-ha area was divided in two paddocks that were randomly assigned to different grazing managements: winter vs. continuous grazing. We randomly selected 22 water run-off paths in each paddock, and in the half of them, we build elongated branch piles. In each run-off path (with and without branch barriers), we recorded the amount of accumulated and lost sediment (during the rainy season), litter biomass, germinable seed bank, richness and cover of plant species, and richness and density of seedlings and saplings of woody species.

Results: Branch barriers promoted sediment accumulation during the first and the second year of the study, depending on grazing management. The temporal and spatial scale of the effect of the branch barriers also depended on grazing management. Branch barriers also trapped litter and seeds, which may have increased the richness and density of seedlings and saplings of woody species.

Conclusions: By intercepting the dominant flow of erosive agents, branch barriers trapped sediment, litter, and propagules of different species. A greater amount of sediment and litter would have improved microsite quality, favouring seed germination and seedling emergence of tree and shrub species, which are key to maintain and/ or reconstitute the structure and composition of the forest community in the long term. Therefore, redistributing biological legacies at strategic locations can be a useful and cost-less retention forestry practice to be applied in multipurpose forest management and conservation strategies.

Keywords: Branch barriers, Ecosystem degradation, Multipurpose forests, Soil erosion, Winter grazing

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Background

Forests play a key role in the livelihoods of rural people around the world. It has been estimated that forests provide fuel, building materials, food, and medicine for more than a billion people (FAO 2018). Forest use is so widespread that the vast majority of the world's forest surface (e.g. approximately 85% of the global forest area) is managed for provision of goods and/or services (FAO 2010). However, traditional use practices can have major negative impacts on forests, by promoting their structural and functional simplification (Thompson et al. 2011; Lindenmayer et al. 2012).

Firewood extraction, timber harvesting, and cattle raising are the main productive activities sustained by forests, being also the main causes of their degradation. Forest overuses can lead to their structural-functional simplification, which is mainly driven by the loss of soil cover. Increasing proportions of soil exposed to erosive agents, as water and wind, can induce resource loss at soil level (Pimentel 2006). This is because erosion not only causes the loss of structuring soil particles, but also nutrients, litter, organic matter, seeds, and beneficial microorganisms, among others, decreasing in turn, soil moisture, and water infiltration capacity. Therefore, soil erosion can drastically decrease forest's productivity (Pimentel and Kounang 1998), and thus, their ability to provide many environmental goods and services.

Mitigation of soil erosion in multipurpose forests can be achieved by applying retention forestry practices (Gustafsson et al. 2012). Retention forestry is the provision for continuity in structural, functional, and compositional elements over forest generations. Thus, some elements of the forest (e.g. vegetation structures, organisms, and patches) are retained in the long term, enriching and adding complexity to the managed forest (Lindenmayer et al. 2012). In this sense, managing the biological legacy may be used as a retention forestry practice (Franklin et al. 2000). This is because retention forestry has emerged from the recognition that even intense and/or extensive natural disturbances, that kill trees and modify ecosystem functioning, leave biological legacies (e.g. standing dead trees, downed tree boles, plants, fungi, etc.) and spatial heterogeneity in the post-disturbance system (Gustafsson et al. 2012). In silvopastoral systems of Arid Chaco, huge amounts of woody debris are generated as a result of the usual forest clearing practices carried out to: (i) extract firewood, (ii) increase the surface for forage production, and (iii) facilitate cattle access to forage growing in the understorey. These woody debris may be burned in situ or may remain sparse in the field without any specific function or location. However, they can be useful if they are grouped together or relocated at strategic points in the landscape. Thus, retaining and redistributing this biological legacy at strategic locations can create sediment, litter, and seed sinks (Ludwig and Tongway 1996; Tongway and Ludwig 1996) which may prevent, stop, or even reverse erosive processes. In this study, we assessed if redistributing woody debris in the form of branch barriers located perpendicular to water run-off paths can slow, or even stop, soil water erosion, and thus, resource run-off in multipurpose forests of Arid Chaco (Argentina).

The forests of Arid Chaco have undergone severe degradation processes (Montenegro et al. 2005), which significantly decreased their productivity (Torrella and Adámoli 2005). In the last four decades, 71% of the area previously covered by forests was degraded. Specifically, vegetation cover changed from closed forest to shrubland (i.e. approximately a 48.9%) or to open forest (i.e. near 22.1%) in the southeastern portion of Arid Chaco (Córdoba, Argentina; Hoyos et al. 2013). Indeed, only a 6.5% of the area detected as forests in 1979 remained as this cover type in 2010 (Hoyos et al. 2013). These remnants are fragmented and immersed in a matrix composed by patches with different degradation degrees, from secondary forests, standing dead forests, shrublands, and grasslands, to lignified savannahs and degraded open-lands (Carranza and Ledesma 2009). Forest degradation was mainly caused by a period of excessive timber harvest for railways construction (from 1900 to 1950) which was followed by a period of overgrazing and firewood extraction extending until present (Cabido et al. 1994; Silvetti 2012). The main consequences of such overuse are water and wind erosion (Torrella and Adámoli 2005). In this situation, there is an urgent need to manage forests with the objective of avoiding further degradation, and retention forestry could be used as an alternative to the above-mentioned traditional use practices. The aim of this study was to stop and even reverse soil erosion by constructing branch barriers that were expected to obstruct water flow, avoiding resource run-off. The hypothesis was that these obstructions would capture resources entrained in water flows, thus creating resource sinks, adding heterogeneity and complexity to the managed forests. We used a silvopastoral system of Arid Chaco (Argentina) as a case study to test this hypothesis. Specifically, we tested the effect of branch barriers on sediment gains and losses, litter accumulation, germinable seed bank (richness, diversity, and density of emerged individuals), richness, and cover of herbaceous and woody species, as well as richness and density of seedlings and saplings of woody species. Also, the effect of branch barriers on resource dynamics was tested under two grazing management practices: seasonal grazing management (e.g. stocking rate adjusted to annual forage availability and delay of grazing until winter season) vs. traditional grazing management (e.g. grazing throughout the year with fixed stocking rates which are decoupled from the annual forage availability) (Quiroga et al. 2009). This is the first study that evaluates the efficiency of a retention forestry practice under different grazing managements in the avoidance or reversal of degradation processes associated with soil erosion.

Methods

Study area

Arid Chaco is located in northwestern Argentina (between 28°15′ and 33°30′ S, and between 64°30′ and 67°31′ W) and covers approximately 10 million ha across La Rioja, San Juan, Cordoba, San Luis, Santiago del Estero, and Catamarca provinces (Morello et al. 1985; Karlin et al. 2013). This region has a subtropical semiarid climate, with hot summers and mild winters. January is the month with highest average temperature (26 °C), and July is the coldest (11 °C) (Prohasca 1959). Mean annual precipitation decreases from 500 mm in the east to 300 mm in the west, with 80% falling in late spring and summer seasons (Morello et al. 1985; Cabido et al. 1993, Karlin et al. 2013). Precipitations are usually torrential, with great intensity and low frequency. Therefore, in degraded areas, precipitations are the main erosive agent, because the raindrops affect bare soil and drag sediments to the lower areas of the landscape (Karlin 2012). Soils are aridisols and entisols with local texture variations (Gomez et al. 1993). The characteristic vegetation of this region is a low xerophytic forest (Cabido et al. 1994). Dominant tree genera include *Aspidosperma* and *Prosopis*. Dominant shrub genera are *Larrea*, *Mimozyganthus*, *Senna*, *Capparis*, *Vachelia*, and *Celtis*, while the herbaceous layer is usually dominated by perennial C4 grasses of the genera *Trichloris*, *Chloris*, *Pappophorum*, *Aristida*, and *Setaria* (Ragonese and Castiglioni 1970; Morello et al. 1985).

The study was conducted in the southeastern portion of Arid Chaco region, more specifically in the foothills of Pocho western hills of Córdoba province (Fig. 1). In this area, soils are sandy/sandy-loam, dominated by Torripsamment ustico, excessively drained and susceptible to water and wind erosion (Gorgas and Tassile 2003). Well-conserved forests have an open tree layer dominated by *A. quebracho-blanco*, with a maximum height of 15 m, accompanied with sparse individuals of *P. flexuosa* and *Zizyphus mistol*. The shrub layer can reach up to 4 m in height and can be dominated by *M. carinatus*, *C. pallida*, and *L. divaricata*. Herbaceous layer is usually dominated by perennial grasses, e.g. *T. crinita*, *T. pluriflora*, and *Gouinia paraguayensis* (Cabido et al. 1994).

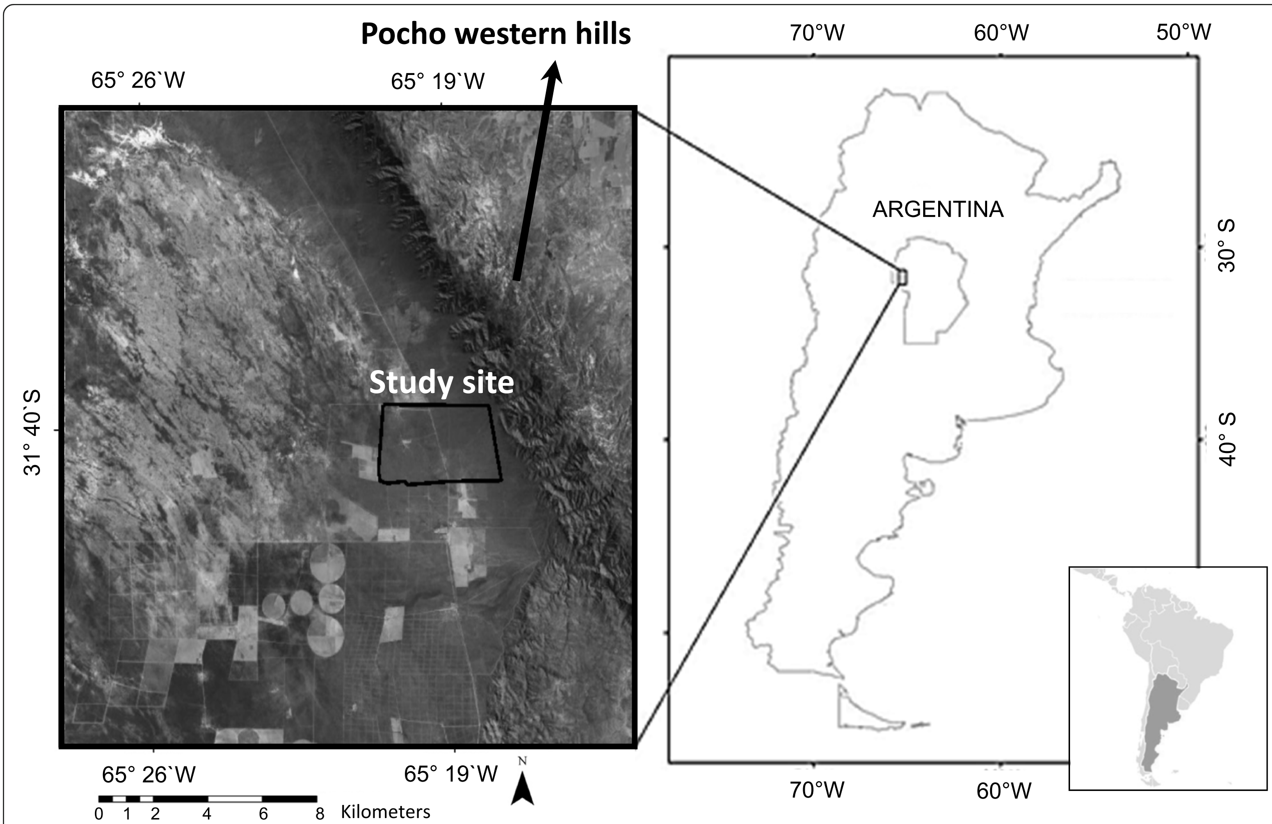

Fig. 1 Location of the study site (black polygon) at the foothills of Pocho western hills, Córdoba province, Argentina

Forest physiognomy and composition can change to shrublands with sparse emergent trees in response to selective logging and livestock raising. In these situations, the tree layer becomes shorter, up to 6–8 m, dominated by *P. flexuosa*, accompanied by scarce emergent and remnant trees of *A. quebracho-blanco*. Shrub layer can be extremely dense, dominated by *M. carinatus* and *C. pallida* (Cabido et al. 1994). Actually, the study area is almost covered by native forests, severely degraded by decades of overgrazing and forestry overuse. Soil erosion evidences are abundant, such as the presence of pedestal plants, furrows or gullies with east-west direction, which are mainly generated by superficial water run-off coming from Pocho western hills during the summer season (Fig. 1).

Experimental design

The study site was established in a multipurpose farm (~ 300 ha) of a small beekeeper and breeder of goats and cattle (31°40′10.11″ S, 65°18′11,5″ W, at 390 m.a.s.l.) (Fig. 1). Within this farm, we selected a 100-ha area that before 1950 supported mixed forests of *A. quebracho-blanco* and *P. flexuosa*. Actually it is a shrubland generated by decades of livestock overgrazing and forestry overuse. This area was selected because it is representative of most multipurpose farms of the region. We manipulated two factors in the experiment: (i) grazing management with two levels (winter grazing vs. continuous grazing) and (ii) branch barriers with three levels (branch barriers, upstream of branch barriers, and without branch barriers) nested within grazing management. In the selected area, we fenced a 1-ha paddock (October 2010) with the aim to seasonally manage livestock grazing and annually regulate the stocking rate. Due to logistic and budgetary limitations, we only could build one fenced paddock. Specifically, this paddock consisted in a temporary enclosure in which grazing was excluded until winter season, while it was grazed with a stocking rate adjusted to annual forage availability (hereafter winter grazing). Forage availability was assessed annually in autumn (May), by systematically harvesting standing grass biomass. Grass standing biomass was estimated by clipping all forage plants growing within 0.75 m^2 frames at 5-m intervals in six transects of 85 m disposed in east-west direction (76.5 m^2). Grass standing biomass was clipped at 10 cm above soil level and collected in nylon bags. The harvested forage biomass was oven dried at 60 °C until constant weight (kilogrammes of dry matter per hectare, hereafter kg DM ha^{-1}). Before grazing, the average forage availability in this paddock varied annually between 747 and 803 kg DM ha^{-1}. Stocking rate was adjusted assuming a theoretical consumption of 10 kg DM day^{-1} by each animal unit. An animal unit (AU) is the annual average of the forage requirements of a 400-kg weight cow, that breeds and rears a calf until the weaning (e.g. at 6 months of age with 160-kg weight) including the forage consumed by the calf. This AU can also be equivalent to the forage requirements of a 410-kg weight steer with a daily weight increase of 0.5 kg (Attwood and Heavey 1964; Cocimano et al. 1973; Holechek 1988; National Academies of Sciences, Engineering, and Medicine 2016). Livestock remained in the paddock until the biomass of palatable forage species was consumed until 10 cm above soil level. Considering a 50% of efficiency of forage harvesting, two cows remained grazing in the paddock during a time period that varied annually from 20 to 30 days, depending on forage availability. Adjacent to the fenced paddock, we delimited a 1-ha paddock that remained open to grazing throughout the year with a stocking rate of 0.16 AU ha^{-1} (hereafter continuous grazing). It must be noted that this stocking rate is greater than 0.1 AU ha^{-1} which is the usual stocking rate of Arid Chaco (Quiroga et al. 2009). Forage availability was also assessed in the paddock under continuous grazing through the procedure explained above. However, because forage availability was estimated while the paddock was being grazed, the permanent consumption of forage by cattle does not allow the accumulation of forage biomass. Therefore, in continuously grazed paddock, mean forage availability varied annually between 10 and 31 kg DM ha^{-1}.

After that, we randomly selected 11 pairs of water run-off paths (December 2010) with similar depth and width in each paddock (e.g. 22 run-off paths in winter grazing paddock and 22 run-off paths in continuous grazing paddock, $n = 44$). Run-off paths were randomly assigned to the barrier treatments: with and without branch barriers. To decrease water speed and sediment drag in each paddock, we built 11 elongated branch barriers perpendicular to each one. Branch barriers were built with firewood and woody debris stacked in form of elongated piles ~ 1-m long, 0.5-m width, and 0.3-m height. To be able to record sediment gains and losses, we used iron stakes buried in the ground with the upper end exposed 150 mm above soil level. Considering water flow direction, iron stakes were placed in the upstream side of branch barriers (hereafter branch barriers), 3 m upstream the branch barriers along the same water run-off path (hereafter upstream) and in run-off paths without branch barriers (hereafter control). Upstream iron stakes were placed with the aim to test the spatial scale at which branch barriers affect sediment deposition upstream (e.g. close to 3 m). In total, 66 iron stakes were installed: (i) 33 within the fenced paddock with regulated stocking rate and winter grazing and (ii) 33 in the nearby paddock with traditional stocking rate and grazing throughout the year. Within each paddock, 11 iron stakes were installed for each treatment: (i) 11 in branch barriers, (ii) 11 upstream of branch barriers, and (iii) 11 in control run-off paths.

Sampling procedure

To estimate the amount of sediment gained or lost in water run-off paths under different treatments from 2010 to 2014, we measured the exposed portion of each iron stake with a digital calliper during the wet season (December to February). Every year, before the beginning of the wet season, all iron stakes were flushed at 150 mm above soil level. The exposed portion of iron stakes was measured after single or accumulated rain events of ~ 30 mm. Because cattle may trample or flip the stakes, to reduce the likelihood of losing data, we performed several measurements over each stake along each wet season. For this sampling, the experimental unit consisted in a plot divided into three sub-plots, each containing one of the three levels of the branch barrier factor (e.g. branch barrier, upstream, and control; $n = 11$ plots in each paddock).

To assess if branch barriers can serve as litter and seed-sinks providing refugee to seedlings and saplings, we characterized the microsites created by each branch barrier in comparison with control run-off paths. We recorded the following variables 5 years after the addition of branch barriers (September 2016) in each paired run-off path: (i) cover and richness of perennial species < 2-m height, (ii) seedling and sapling density of woody species, (iii) litter biomass, and (iv) germinable seed bank. The experimental unit for this sampling consisted in a plot divided into two sub-plots, each containing one of two levels (branch barrier and control with 11 plots in each paddock).

The cover and richness of perennial species and the density of seedlings and saplings of woody species were sampled in 1-m^2 plots (Nicotra et al. 1999). Plant cover and richness were recorded for herbaceous and woody species < 2-m height (e.g. individuals susceptible to livestock browsing). Woody individuals < 0.3 m with cotyledons or cotyledonal scars were considered as seedlings, whereas those higher than 0.3 m but shorter than 1.5 m were considered as saplings (Brassiolo et al. 1993).

Litter was collected in each run-off path by using a 0.4 × 0.4-m square. To collect litter samples underneath branch barriers, woody debris were carefully removed. Litter samples were oven-dried for 72 h and their biomass was estimated by using a precision scale (ACCU-LAB®, USA).

To estimate the germinable seed bank, we collected three soil sub-samples (13-cm diameter and 3-cm depth) in each run-off path. Soil samples were collected within the same 1-m^2 plot used for vegetation sampling, but were not overlapped with the area in which litter samples were collected. Thus, litter was included in each soil sub-sample. Soil samples were stored in nylon bags in a dark room until processing. In total, we gathered 44 soil samples composed by three sub-samples each. Soil samples were sieved (October 2016) through a 1.3-cm wire mesh to remove organic debris and stones. Sieved samples were distributed in pots of 3-cm depth, which were placed in a glasshouse with uncontrolled conditions for 9 months until seedling emergence had ceased (Franzese et al. 2016). Pots were watered regularly and seedlings were taxonomically identified, counted, and removed every week.

Data analysis

To test the effect of branch barriers on sediment deposition dynamics (gains and losses), we calculated for every year the sediment accumulated during the wet season. Because sediment accumulation was recorded several times within the summer season (e.g. after single or accumulated rain events of ~ 30 mm), summer sediment accumulation was calculated as follows:

$$\text{Summer sediment accumulation}_{(2010\ldots2014)} = \sum\nolimits_{\text{Date 1}}^{\text{Date } n} \text{accumulated sediment}$$

where

$$\begin{aligned}\text{Accumulated sediment}_{\text{Date } 2\ldots n} &= \text{length of iron stake above soil level}_{\text{Date } 1\ldots n-1} \\ &\quad - \text{length of iron stake above soil level}_{\textit{Date } 2\ldots n}\end{aligned}$$

Because rainfall varied annually (Table 1), summer sediment accumulation was relativized by the rainfall millimetre accumulated during the measurement period, allowing the unbiased between-year comparison of sediment deposition dynamics:

$$\begin{aligned}&\text{Accumulated sediment per fallen mm}_{(2010\ldots2014)} \\ &= \text{summer sediment accumulation}_{(2010-2014)} \\ &\times \left(\text{mm of rain fallen during measurement period}_{(2010\ldots2014)}\right)^{-1}\end{aligned}$$

This variable was calculated for summer 2010–2011, 2011–2012, 2012–2013, and 2013–2014. To assess the net effect of treatments at the end of the study period, we calculated the net summer sediment accumulation as follows:

$$\text{Net summer sediment accumulation} = \sum\nolimits_{2010}^{2014} \text{summer sediment accumulation}$$

Finally, to test if branch barriers can trap litter and seeds creating suitable microsites for plant recruitment, we calculated the following response variables: litter biomass m^{-2}, richness, diversity, and density of seedlings emerged from soil seed bank, cover and richness of herbaceous and woody species, and density of seedlings and saplings of woody species.

Table 1 Annual rainfall recorded in the study area during the wet season

Summer (year)	Rainfall (mm)	Mean rainfall event ± SD (mm)	Min and max rainfall event (mm)
2010–2011	165	27.0 ± 19.0	05/60
2011–2012	490	54.4 ± 55.1	10/180
2012–2013	245	23.7 ± 13.5	07/50
2013–2014	528	28.1 ± 10.5	15/40

We fitted a linear mixed model (LMM) to test the effect of grazing management and branch barriers on accumulated sediment per fallen millimetre. Grazing management with two levels (winter grazing vs. continuous grazing), branch barriers with three levels (in branch barriers, upstream, and control), year with four levels (2010–2011, 2011–2012, 2012–2013, and 2013–2014), and the three-way interaction were included as predictors in the LMM. The year was also included as a random factor in the LMM to account for the temporal autocorrelation between repeated measures during four consecutive summers over the same run-off paths and iron stakes. Also, the model accounted for the split-plot experimental design with plot nested within branch barriers, which was nested within grazing management (Crawley 2007). Accumulated sediment per fallen millimetre was included as response variable in the LMM. Significant interactions were tested separately for each summer using Fisher LSD post hoc contrasts. We fitted a LMM to assess the net effect of grazing management and branch barriers on sediment deposition dynamics, including grazing management, branch barriers and the two-way interaction as predictors, and net summer sediment accumulation as the response variable. This model also accounted for the split-plot experimental design with plot nested within branch barriers, which was nested within grazing management (Crawley 2007).

We fitted generalized linear models (GLMs) to test the effect of grazing management and branch barriers at microsite level, including grazing management, branch barriers and the two-way interaction as predictors. Litter biomass, richness, diversity, and density of seedlings emerged from soil seed bank, cover and richness of herbaceous and woody species, and density of seedlings and saplings of woody species were included as response variables in GLMs. The structure of GLMs also accounted for the split-plot experimental design, with plot nested within branch barriers, which was nested within grazing management. The data of soil subsamples were averaged to be able to include in GLMs as a single value for each experimental unit ($n = 44$ plots). The analyses including continuous response variables assumed a Gaussian error distribution and an identity-link function, whereas the analyses including richness and density as response variables assumed a Poisson error distribution and a log-link function. Based on graphical analysis (residuals vs. predicted values), all models satisfied the underlying statistical assumptions, including linearity and the expected relation of the variance to the mean given the nature of the dependent variables error distribution. All models were implemented with the statistical software R version 2.15.1 (R Development Core Team 2012), using the lme4 function (libraries lme4) (Bates et al. 2015).

Results

Branch barriers promoted sediment accumulation, but the temporal and spatial scale of their effect depended on grazing management ($p = 0.003$). Specifically, branch barriers promoted sediment accumulation during the first year in the paddock with winter grazing and during the first and second year in the paddock with continuous grazing (Fig. 2a). Also, the spatial scale of this practice was greater than 3 m upstream of branch barriers in the paddock with winter grazing during the first summer. In contrast, in the paddock with continuous grazing, the spatial extent of sediment accumulation upstream of branch barriers was less than 3 m during the first and greater than 3 m during the second summer (Fig. 2a).

During the first summer, the addition of branch barriers promoted the accumulation of similar amounts of sediments in both paddocks (Fig. 2a). In the paddock with winter grazing, run-off paths with branch barriers accumulated, on average, 55.9 mm of sediment (± 5.9 mm). In relative terms, run-off paths with branch barriers accumulated 158 times more sediment than run-off paths without them (Fig. 1a, $p < 0.001$). In the paddock with continuous grazing, run-off paths with branch barriers accumulated 43.3 mm of sediment (± 9.7 mm). In relative magnitudes, the amount of sediment captured by branch barriers was 17 times higher than that recorded in control run-off paths (Fig. 2a, $t = p < 0.001$). Besides this, in the paddock with winter grazing, the effect of this retention forestry practice was also recorded 3 m upstream of branch barriers. Specifically, iron stakes located 3 m upstream of branch barriers accumulated 44.9 mm of sediment (± 7.6 mm). This sediment amount was 123 times higher than that recorded on iron stakes located in run-off paths without branch barriers (Fig. 2a, $p < 0.001$).

During the second summer, in the paddock under continuous grazing, run-off paths with branch barriers

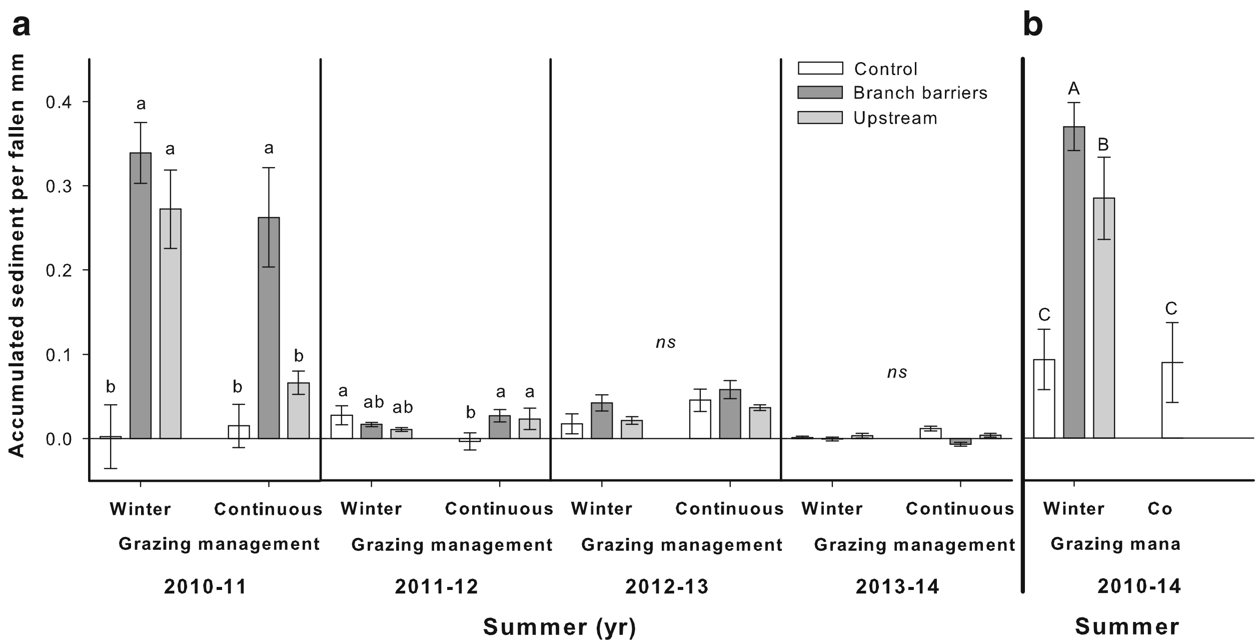

Fig. 2 Mean (± standard deviation, SE) of accumulated sediment per millimetre of rain fallen during the summer season (**a**), and mean (±SE) net summer sediment accumulation during the study period (2010–2014) (**b**) in water run-off paths with branch barriers (branch barriers), 3 m upstream of branch barriers (upstream) and in run-off paths without branch barriers (control); under different grazing managements: winter (a 1-ha temporary enclosure in which grazing was excluded until winter season, when it was grazed with a stocking rate adjusted to annual forage availability), and continuous (a 1-ha paddock that remained open to grazing throughout the year with a stocking rate of 0.16 AU ha^{-1}). Different lowercase letters showed significant differences for grazing management × branch barriers interaction ($p = 0.05$). Different uppercase letters indicated significant differences between levels of one factor ($p = 0.05$)

trapped 13.2 mm of sediment (± 3.5 mm). This amount of sediment was seven times higher than that recorded in run-off paths without branch barriers (Fig. 2a, $p = 0.015$). Also in this paddock, iron stakes located 3 m upstream of branch barriers accumulated similar amounts of sediment (e.g., 11.4 ± 6.1 mm), being six times higher than that recorded in control run-off paths (Fig. 2a, $p = 0.032$).

During the third and fourth summer, sediment gains and losses were similar between treatments in paddocks under different grazing managements ($p = 0.052$ and $p = 0.098$, respectively). It must be noted that sediment deposition dynamics were the most variable in control run-off paths, not only spatially (within year) but also temporally (between years). Specifically, the coefficient of variation (CV) in these run-off paths was greater than in the other treatments, mainly during the first year. Indeed, in that year, the CV of control treatments was among 7–16 times greater than other treatments (Fig. 2a).

At the end of the study, the addition of branch barriers promoted a net sediment accumulation, independently of grazing management (Fig. 2b, $p < 0.001$). On average, the amount of sediments trapped by run-off paths with branch barriers was four times higher than that recorded in run-off paths without them (Fig. 2b; $p = 0.001$). In addition, iron stakes located in run-off paths with branch barriers recorded higher amounts of sediments than that located 3 m upstream of branch barriers (Fig. 2b; $p = 0.021$). Likewise, the net effect of branch barriers reached the scale of 3 m. On average, the amount of sediments recorded in iron stakes located 3 m upstream of branch barriers was two times higher than that recorded in iron stakes placed in control run-off paths (Fig. 2b; $p = 0.018$).

Branch barriers served as litter and seed-sinks. Litter biomass and density of emerged seedlings (from soil seed bank) were greater in run-off paths with branch barriers than in those without them, independently of grazing management ($p < 0.001$ and $p = 0.020$, respectively). Run-off paths with branch barriers accumulated, on average, three times more litter than those without them (Fig. 3a). Also, the density of emerged seedlings was two times greater in run-off paths with branch barriers than in those without them (Fig. 3b).

Overall, 2857 seedlings, belonging to 41 species and 21 genera, germinated from soil seed bank. The 62.7% of the seedlings emerged from soil samples collected in the paddock with winter grazing, whereas the remaining 37.3% of them emerged from soil samples collected in the paddock with continuous grazing. Likewise, the 64.3% of the seedlings emerged from soil samples collected underneath branch barriers, whereas the

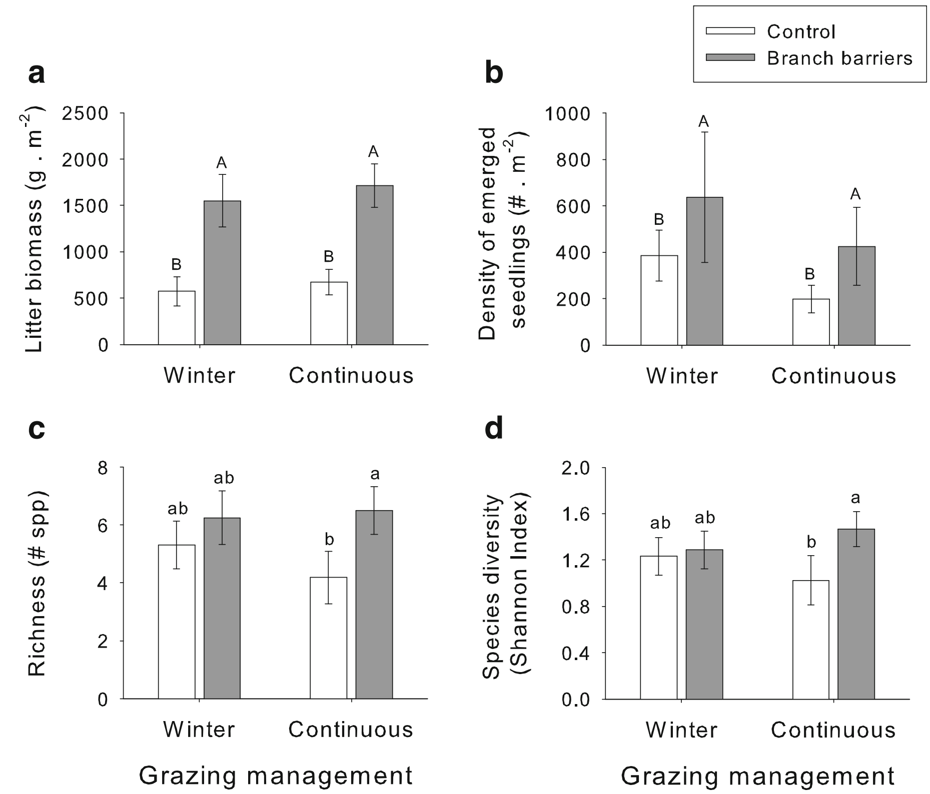

Fig. 3 Mean (± standard deviation, SE) of litter biomass (**a**), density (**b**), richness (**c**), and diversity (**d**) of seedlings emerged from soil seedbank in water run-off paths with and without branch barriers under different grazing managements: winter (a 1-ha temporary enclosure in which grazing was excluded until winter season, when it was grazed with a stocking rate adjusted to annual forage availability), and continuous (a 1-ha paddock that remained open to grazing throughout the year with a stocking rate of 0.16 AU ha^{-1}). Different lowercase letter showed significant differences for grazing management × branch barriers interaction ($p = 0.05$). Different uppercase letters indicated significant differences between levels of one factor ($p = 0.05$)

remaining 35.7% of them emerged from soil samples collected in run-off paths without branch barriers. Two herbaceous species, *Sporobolus pyramidatus* (39.2%) and *Setaria lachnea* (15.7%), accounted for the 54.9% of the emerged seedlings. Only a 0.16% of the emerged seedlings were of woody species: one small tree (*Parkinsonia praecox*) accounting for 0.13% and one shrub (*L. divaricata*) accounting for 0.03%. It must be noted that seedlings of woody species emerged from soil samples collected underneath branch barriers, and 80% of them emerged from soil samples collected in the paddock with continuous grazing (Table 2).

Richness and diversity of germinable seed bank depended on both, grazing management and branch barriers ($p = 0.023$ and $p = 0.007$). Despite richness and diversity of emerged seedlings were greater in run-off paths with branch barriers than in those without them, significant differences between treatments were only detected in the paddock with continuous grazing. Specifically, under continuous grazing, the richness of seedlings that emerged from soil samples collected underneath branch barriers was two times greater than in those collected in control run-off paths (Fig. 3c). Also, species diversity was 43.1% higher in run-off paths with branch barriers than in those without them (Fig. 3d).

Richness and cover of perennial species were similar among run-off paths with and without branch barriers in both paddocks ($p = 0.681$ and $p = 0.454$, respectively). In contrast, seedling density and richness were greater in run-off paths with branch barriers than in those without them, independently of grazing management ($p < 0.001$ for both treatments). Specifically, seedling density of woody species was, on average, five times greater in run-off paths with branch barriers than in those without them (Fig. 4a). Most seedlings were of *A. quebracho-blanco* (81.2%), the main tree species of Arid Chaco forests. We also found seedlings of a small tree and a shrub (*Castela coccinea* and *L. divaricata*, respectively). On average, seedling richness was four times greater in run-off paths with branch barriers than in those without them. We only recorded saplings in the paddock under continuous grazing, whose density was higher in run-off paths with branch barriers than in those without them (Fig. 4b; $p = 0.017$). In fact, we recorded 34 times more

Table 2 Species or genera, family, and relative proportion of seedlings emerged from soil seedbank

Life-form	Species	Family	Relative proportion	Annual/perennial	Native/exotic
Grasses	*Cenchrus ciliaris*	Poaceae	1.88	P	E
	Cynodon dactylon	Poaceae	0.56	P	E
	Setaria lachnea	Poaceae	7.05	P	N
	Setaria leucopila	Poaceae	0.23	P	N
	Sporobolus pyramidatus	Poaceae	39.25	P	N
	Trichloris crinita	Poaceae	15.74	P	N
	Trichloris pluriflora	Poaceae	3.56	P	N
	Unidentified grass	N/A	0.43	–	–
Herbs	*Abutilon* sp.	Malvaceae	1.58	–	–
	Alternanthera pungens	Amaranthaceae	2.73	P	N
	Boerhavia difussa	Nyctaginaceae	0.07	P	E
	Boerhavia sp.	Nyctaginaceae	0.03	–	–
	Euphorbia hyssopifolia	Euphorbiaceae	1.42	A	N
	Euphorbia sp.	Euphorbiaceae	0.1	–	–
	Gomphrena pulchella	Amaranthaceae	7.01	P	N
	Justicia squarrosa	Acanthaceae	1.02	P	N
	Lantana sp.	Verbenaceae	0.26	–	–
	Oxalis sp.	Oxalidaceae	0.2	–	–
	Portulaca confertifolia	Portulacaceae	1.98	P	N
	Portulaca cryptopetala	Portulacaceae	0.43	A	N
	Portulaca oleracea	Portulacaceae	3.26	P	E
	Portulaca sp.	Talinaceae	2.14	–	–
	Selaginella sp.	Selaginellaceae	0.13	–	–
	Solanum juvenale	Solanaceae	0.23	P	N
	Solanum nigra	Solanaceae	0.16	P	N
	Sonchus oleraceus	Asteraceae	2.14	A	E
	Sonchus sp.	Asteraceae	0.01	–	–
	Talinum fruticosum	Talinaceae	0.23	P	E
	Talinum paniculatum	Talinaceae	4.48	P	E
	Taraxacum officinale	Asteraceae	0.3	A	E
	Unidentified sp. 1	N/A	0.03	–	–
	Unidentified sp. 2	N/A	0.07	–	–
	Unidentified sp. 3	N/A	0.03	–	–
	Unidentified sp. 4	N/A	0.03	–	–
	Unidentified sp. 5	N/A	0.23	–	–
	Unidentified sp. 6	N/A	0.1	–	–
	Unidentified sp. 7	N/A	0.07	–	–
	Unidentified sp. 8	N/A	0.64	–	–
Shrubs	*Larrea divaricata*	Zygophyllaceae	0.03	P	N
Woody climber	Unidentified climber	N/A	0.03	–	–
Small tree	*Parkinsonia praecox*	Fabaceae	0.13	P	N

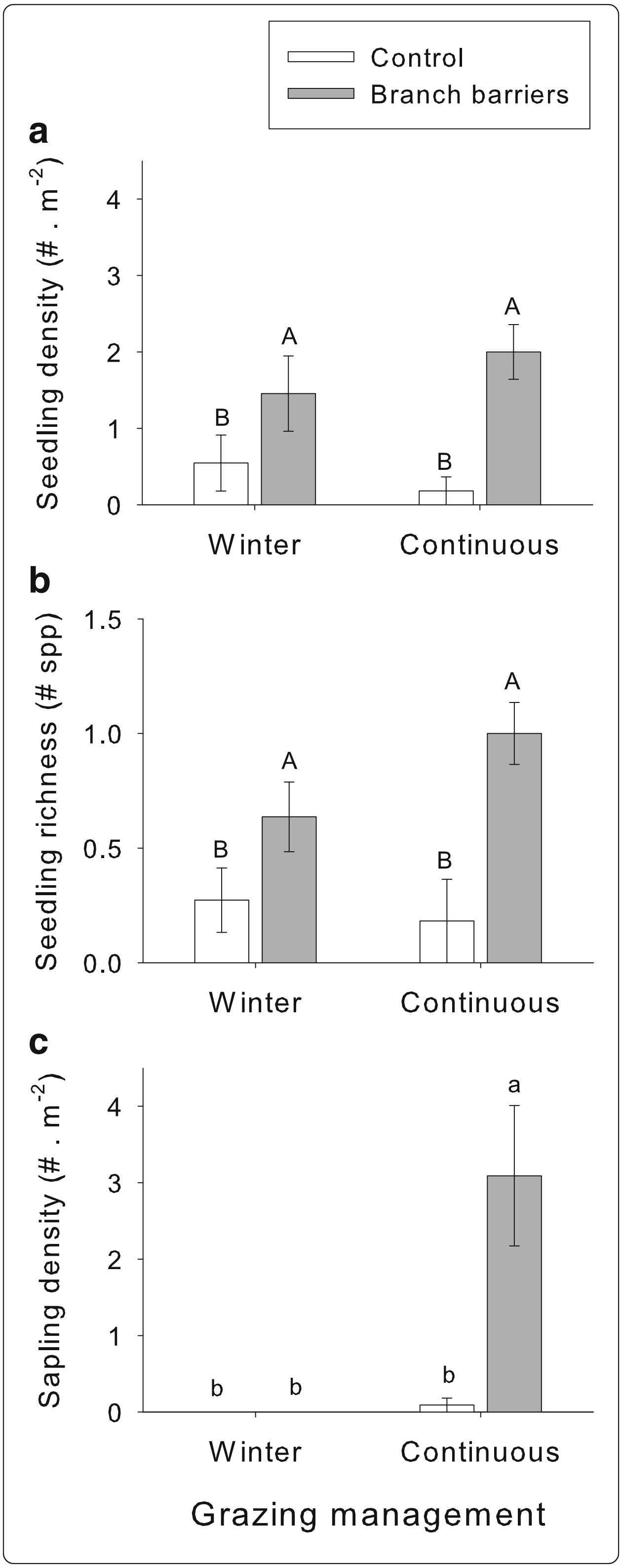

Fig. 4 Mean (± standard deviation, SE) of the density of seedlings (**a**) and saplings (**b**), and richness of seedlings of woody species found at microsite level in water run-off paths with and without branch barriers under different grazing managements: winter (a 1-ha temporary enclosure in which grazing was excluded until winter season, when it was grazed with a stocking rate adjusted to annual forage availability), and continuous (a 1-ha paddock that remained open to grazing throughout the year with a stocking rate of 0.16 AU ha^{-1}). Different lowercase letter showed significant differences for grazing management × branch barriers interaction ($p = 0.05$). Different uppercase letters indicated significant differences between levels of one factor ($p = 0.05$)

saplings in run-off paths with branch barriers than in those without them (Fig. 4c). *A. quebracho-blanco* was the most abundant species accounting for the 88.8% of the saplings. The remaining 11.2% of the saplings were of *P. praecox* a small tree species of Arid Chaco forests.

Discussion

Retaining and redistributing biological legacies reduced the rate of resource loss by erosive agents in a degraded silvopastoral system. In this sense, branch barriers acted as sinks of sediments (Fig. 2), litter, and seeds (Fig. 3), creating suitable microsites for seed germination and seedling establishment (Fig. 4). Also, this practice added complexity to the soil surface of the managed forest, thereby, decreasing the extent and degree of connectivity among patches of bare ground (López et al. 2013; Tongway and Ludwig 1996; Ludwig et al. 2005; Kimiti et al. 2017). Therefore, retaining a portion of the biomass extracted by the usual forest clearing practices and redistributing it at strategic locations on the landscape can be used as a retention forestry practice to improve the functionality of the forest being managed for goods and services provision (Fig. 5b).

The temporal and spatial scale of the effect of branch barriers on sediment accumulation depended on grazing management. Specifically, branch barriers promoted sediment accumulation during the first year in the winter grazed paddock and during the first and second year in the continuously grazed paddock (Fig. 2a). Although the spatial reach of this practice was the same in both paddocks (e.g. > 3 m upstream of branch barriers), sediment accumulation took 1 year to reach this scale in the paddock with winter grazing, whereas 2 years in the paddock with continuous grazing (Fig. 2). Also, during the second year of the study, run-off paths without branch barriers from the winter grazed paddock accumulated similar amounts of sediment than run-off paths with branch barriers of the continuously grazed paddock (Fig. 2). These differences, regarding the temporal and spatial scale of the effects of branch barriers, could be attributed to the deferral of grazing until winter season and to the adjustment of the stocking rate to annual

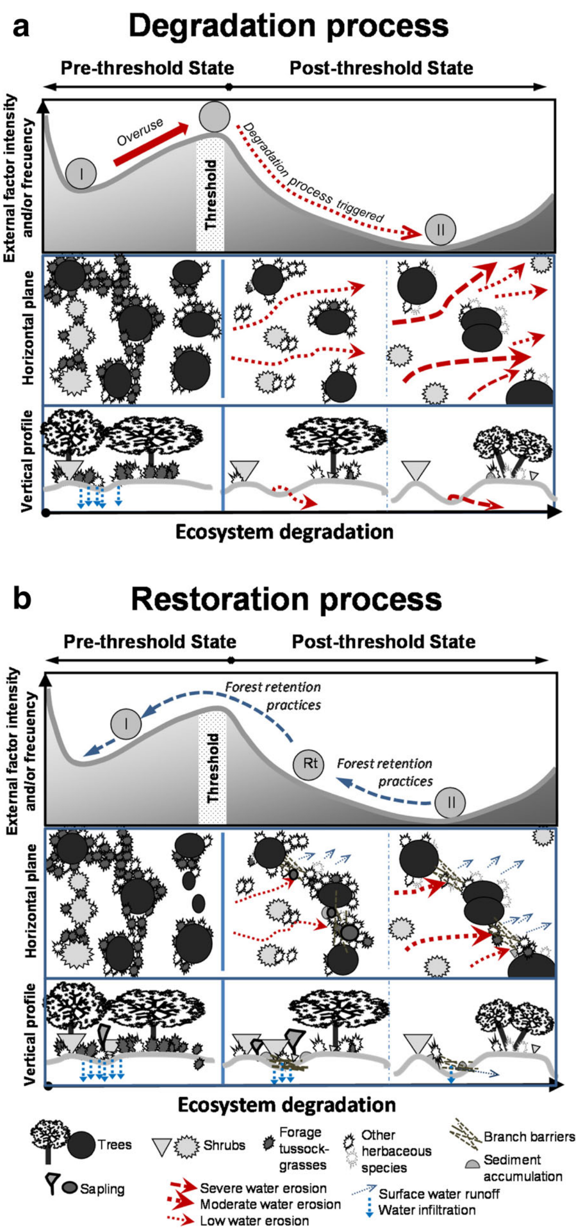

Fig. 5 Hypothetic schemes of degradation (**a**) and restoration (**b**) processes for xerophilous forests. Three sub-figures are shown in each panel (**a**, **b**): at the top, there is a cup-ball scheme illustrating a threshold between alternative states (grey balls I and II); in the middle, the system is illustrated in a horizontal plane; whereas at the bottom, the system is illustrated as a vertical profile. In **a**, decades of livestock-forestry overuse causes a significant reduction in the plant cover of a hypothetic ecosystem (ball I), triggering soil erosion processes (e.g. soil, organic material, and water loss). Consequently, the system cross a threshold to an alternative state (ball II). In **b**, the addition of branch barriers would trigger ecosystem recovery (e.g. transition of ball Rt between state II to state I). Branch barriers obstruct or reduce water run-off (increasing soil infiltration), increasing the retention of sediment, organic matter (e.g. litter) and seeds, generating favourable microsites for the emergence and survival of new tree individuals (e.g. seedlings and saplings)

forage availability. In this sense, this grazing management has several benefits over traditional grazing management. First, winter grazing has the benefit of maintaining the soil covered with the herbaceous layer when the likelihood of resource loss by run-off processes is highest (i.e. due to the high intensity of rain events during summer season, Karlin 2012), minimizing resource loss during this period. In contrast, continuous grazing maintains a low soil cover throughout the year, because cattle permanently consumes biomass of the herbaceous layer, even in the rainy season. Second, when paddocks are grazed during winter, the regeneration of the biomass consumed by cattle occurs during summer, which is also the period of maximum vegetative growth, and when water availability is also the highest. In contrast, in continuously grazed areas, the regeneration of the consumed biomass must occur at any time of the year, lowering the efficiency of forage production (Quiroga et al. 2018). Third, the adjustment of the stocking rate to annual forage availability avoids the depletion of root reserves, allowing the regrowth of forage biomass in the subsequent growing season. Fourth, when paddocks are grazed during winter, the soil remains covered during summer, having the additional benefit of buffering harsh micro-environmental conditions that occur during that period. On the contrary, in areas continuously grazed, the high temperatures of summer season together with a lower soil cover, given by the permanent grazing, cause the loss of soil moisture by direct evaporation, increasing harshness of micro-environmental conditions. Finally, keeping the ground covered during the rainy season has the benefit of decreasing the speed and kinetic energy of water run-off, favouring their infiltration into the soil profile. These benefits together may increase the rain use efficiency of the system for forage production and reserve storage. According to our results, winter grazing decreased the rate of resource loss by erosive agents, and in combination with branch barriers, may have caused a synergistic effect that avoided the resource run-off at a spatial scale greater than the addition of branch barriers without a specific grazing management (Fig. 2). This is because iron stakes located in control treatment within winter grazing paddock and those located in branch barriers treatment outside winter grazing paddock recorded similar amounts of sediments during the second year of the study (Fig. 2). Consequently, the temporal retention of herbaceous cover, given by the delay of cattle grazing until winter season and the adjustment of the stocking rate to annual forage availability, can also be considered as a retention practice that can be applied in silvopastoral systems.

The decrease of soil cover may promote the loss of resources at soil level, and their flux towards other parts of the landscape, and thus, any practice aimed to reduce or reverse resource loss could avoid the degradation of forests employed for provision of goods and services (Ludwig and Tongway 1996; Simons and Allsopp 2007) (Fig. 5). The retention of the biological legacies and their re-distribution in the form of elongated branch piles at strategic locations, promoted sediment accumulation in both paddocks (Fig. 2). Branch barriers covered the soil and offered a physical barrier to water flow, which may have decreased the kinetic energy of raindrops and their subsequent impact on the soil, as well as the speed of superficial water run-off reducing, in turn, resource loss (Myronidis et al. 2010; Fig. 5). During the first year of the study, branch barriers accumulated at least 40 mm of sediment in water run-off paths from both paddocks. Other authors found the same pattern, but with different magnitudes (Tongway and Ludwig 1996; Kimiti et al. 2017). In the semiarid woodlands of eastern Australia, Tongway and Ludwig (1996) recorded that branch piles promoted the accumulation of 0.6-mm sediment per year. Also, in semiarid savannahs of eastern Africa, Kimiti et al. (2017) recorded 25 mm of sediment accumulated annually upstream of branch barriers. Differences in effect magnitude between studies can be attributed to differences between study regions regarding to (i) terrain slope; (ii) intensity, frequency, and duration of precipitation events; (iii) degree of soil cover; and (iv) soil texture, which mainly determine the kinetic energy of water flows and the susceptibility of soil particles to be dragged away. Kimiti et al. (2017) also recorded the extent of soft sediment that covered the hard-capped soil on the upslope side of branch barriers. They found that the average spatial reach of this practice was 4.7 ± 0.87 m, a similar scale that we found in our study (e.g. > 3 m). These results suggest that branch barriers can slow the rate of resource loss (e.g. sediments, litter, and water) at a local scale (Fig. 2).

Branch barriers stabilized sediment movement within the landscape. Sediment gains and losses were much more variable in run-off paths without branch barriers than in those with them (e.g. CV of control treatments during the first year of the study was 7–16 times greater than the other treatments; Fig. 2a). Probably, the control treatments recorded greater variability than the other treatments because they were vulnerable (e.g. higher exposure) to factors that can vary from year to year at microsite level. For example, temporal changes in microtopography, such as the fall of a branch over some gully, disturbances on the soil surface caused by micro- and meso-fauna, among others.

The addition of branch barriers may enhance microsite quality through different mechanisms. On one hand, covering the soil with branches can reduce the soil exposure to solar radiation and wind, decreasing thermal amplitude and potential evaporation at the soil surface, and thus, increasing soil moisture (Castro et al. 2011; Hanke et al. 2011), the rate of water infiltration, and soil respiration (Tongway and Ludwig 1996) (Fig. 5b). On the other hand, branch barriers can promote litter accumulation (Fig. 3a), and this effect was recorded 5 years after the addition of these protective structures to the forest floor. It has been documented that litter deposition can also improve micro-environmental conditions and may even increase soil fertility. This is because the presence of litter also reduces the incidence of solar radiation and decreases temperature and evaporation at the soil surface, having the additional benefit of providing significant amounts of nutrients through their decomposition processes (Dormaar and Willms 1992) (Fig. 5). Therefore, the addition of branch barriers and the subsequent litter deposition can have a positive synergistic effect at microsite level, generating favourable conditions for seed germination, seedling emergence, and establishment.

The richness, diversity, and density of seedlings that emerged from soil seed bank (Fig. 3b–d), the richness and density of seedlings (Fig. 4a, b), and the density of saplings (Fig. 4c) of woody species were significantly higher in run-off paths with branch barriers than in those without them. These results are in accordance with other studies that found higher cover of ephemeral and annual herbaceous species (Simons and Allsopp 2007; Kowaljow and Rostagno 2013) as well as perennial grasses (Ludwig and Tongway 1996; Kowaljow and Rostagno 2013) in treatments with piles of branches in comparison with other treatments. Surprisingly, we did not find saplings in the winter grazed paddock (Fig. 4c). This could be because grass species may outcompete saplings of woody species (Grime 2001; Scholes and Archer 1997; Rusch et al. 2017) or because of apparent competition between grass species and saplings (e.g. saplings spatially associated with the herbaceous layer were more easily browsed than saplings growing in environments with low herbaceous cover) (Holt 1977; Wada 1993; Burger and Louda 1994; Holt and Lawton 1994). Overall, we found that by intercepting the dominant flow of erosive agents, branch barriers trapped propagules that usually are transported by water or wind (Fig. 5b). The accumulation of a rich and diverse seed bank mainly of herbaceous species (Table 2) suggests that this practice would provide temporal continuity in the composition of the herbaceous layer, which is of high value in silvopastoral systems. Also, a greater amount of sediment and litter would have improved microsite quality underneath branch barriers, favouring not only the recruitment of forage species, but also tree (*A. quebracho-blanco*, *P. praecox*, *C. coccinea*) and shrub (*L. divaricata*) species (Fig. 4). Recruitment of tree species is essential to maintain and reconstitute the structure and composition of the forest community in the long term (López et al. 2011; Cavallero et al. 2015; Peri et al. 2017a).

The cover and richness of herbaceous and woody species were similar in water run-off paths with and without branch barriers. Regarding the woody species, lack of differences between treatments could be due to the slow growth rate of Arid Chaco trees (Perpiñal et al. 1993; Juárez de Galindez et al. 2006). This may cause a very slow response to the addition of branch barriers, which could not be detected in a 5-year period. Also, the lack of differences between treatments may be attributed to the fact that branch barriers would not provide an efficient protection against browsing to trees > 0.3 m. Therefore, more studies must be conducted to disentangle the mechanisms underlying these results. If browsing is the factor that limits the persistence of individuals > 0.3 m, branch barriers should be covered with a loose shrub brush (thorny fine branch material) aimed to stop cattle damage. This design was probed to be efficient in animal deterrence in other studies (Kimiti et al. 2017).

Implications of the study for the management of multipurpose forests

The results presented here are derived from a case study including one replicate for the factor associated with grazing management and 11 replicates of branch barriers, for each grazing situation. Thus, the inferences that can be made about our results are more robust at microsite scale than at paddock scale (because branch barriers were replicated within each grazing situation). Specifically, inferences based on the response variables that recorded a significant interaction between grazing management and branch barriers should be interpreted carefully to avoid erroneous extrapolations to other systems. In this sense, to be able to make more precise inferences, more studies should be done on the effect of the addition of branch barriers in silvopastoral systems facing different grazing managements. Nevertheless, branch barriers promoted the accumulation of sediments, litter, and seeds in both grazing situations. This result suggests that the addition of branch barriers can be used as a retention forestry practice in xerophilous forests that face degradation drivers and biophysic limitations similar to the system that we studied. This can provide the benefit of decreasing resource loss and thus increase the efficiency of their maintenance into the managed system.

Although our study was performed at a paddock scale, it can be scaled up at a stand scale. This is because the management practices, either forestry or livestock, can generate large amounts of woody debris. For example, after timber and firewood harvesting operations, branches with diameter < 5 cm are left in the forest. This kind of woody debris can also be generated by pruning practices in timber stands. Specifically, in a *Prosopis alba* experimental plantation from Santiago del Estero (Argentina) with densities from 450 to 4500 trees ha^{-1}, the pruning practices during three consecutive years, generated between 3.1 and 9.8 tn DM ha^{-1} of woody debris (Zarate 2017). In silvopastoral systems implemented in the Chaco region, the great majority of trees and shrubs with DBH < 10–15 cm are removed, crushed, and left as debris (Kunst et al. 2016). The huge amounts of woody debris generated by these practices usually remain in the stands without any specific function or location and can naturally degrade or burnt in situ (increasing the risk of wildfires). However, if a specific function and location is assigned to the woody debris, they can contribute to prevent, reduce and even stop degradation processes that can be triggered by the decrease in soil cover. In addition, woody debris go through a long decomposition period because of their high amounts of lignin, having the benefit of remaining in the harvested stand until the next rotation. Consequently, we believe that the proposed retention forestry practice can be implemented at the stand scale without associated costs or additional management efforts.

This approach can also be applied to other forest legacies that are generated by the implementation of silvopastoral systems in the Chaco region. Specifically, the most common intervention, to facilitate cattle access to forage biomass growing in the understorey, is the removal of a percentage of the shrub layer. Woody individuals, especially trees with DBH > 10–15 cm, are left standing in different patterns and densities (Kunst et al. 2016). These management aims to decrease light, water, and space competition between herbaceous and shrub layers, thus increasing forage productivity. However, many of the woody species have the ability to re-sprout, so these interventions must be recurrent (2–3-year interval) to avoid the recovery of the shrub layer, ensuring the maintenance of a profuse herbaceous layer (Borrás et al. 2017; Peri et al. 2017b). This kind of interventions on the forest leaves other biological legacies than woody debris, which can also play important roles in the managed forests. The trees that are left standing act as pollen, nectar, and seed sources (increasing genetic diversity and ensuring the temporal continuity of the tree layer); provide shade and habitat for other plants (epiphytic or parasitic), fungi, and animals; provide nutrients through the decomposition of their litter; and retain soil and host microorganisms through their roots. The shrub layer also can perform these functions and additionally can provide refuge to wildlife that pollinate flowers and disperse seeds of the main tree species, as well as can protect tree seedlings and saplings against livestock herbivory. Therefore, to be able to retain other biological legacies than woody debris, the interventions to the understorey must be carefully planned and should be aimed to maintain not only large trees, but also intact forest patches which include the shrub layer.

The implementation of retention forestry in silvopastoral systems should consider the management of biological legacies at distinct spatial and temporal scales. On one hand, forestry management should be planned at a stand scale, which usually varies in space from tens to hundreds of hectares, and harvest rotation is usually planed in terms of several decades. For this management scale, leaving standing trees as sources of pollen, nectar, seeds, and thus, genetic diversity can be implemented as a retention forestry practice. On the other hand, livestock management should be planned at a narrow scale, which usually varies spatially in terms of tens of hectares, and temporally from 2 to 5 years (in the case of the recurrent removal of the shrub layer) to several months (in the case of the seasonal livestock rotation for grazing at different paddocks). For this management scale, it is essential to leave at least 50% of intact forest patches, which provide refugee for pollinators and seed dispersers, as well as refuge for tree seedlings and saplings against browsing (Borrás et al. 2017; Peri et al. 2017b). This has the additional benefit of increasing the functional diversity and redundancy of the managed ecosystem (Díaz et al. 2007; Easdale and López 2017). The woody debris generated by silvopastoral practices, either forestry or livestock raising, can be retained and redistributed at strategic landscape locations at both management scales, having the benefit of increasing soil cover and mainly interrupting the flow of erosive agents at soil surface.

Conclusions

Some resources as water, organic matter, and nutrients are limited in arid and semiarid ecosystems, and the loss of plant cover significantly decreases the efficiency with which these resources are maintained in the ecosystem. Therefore, silviculture interventions should be aimed to enhance resource maintenance or even capture the lost resources, may avoid further degradation in forests under anthropic use, or even trigger the recovery of degraded forests. The retention of biological legacies allows to maintain or even capture resources that otherwise would be lost from the ecosystem, by flowing away through other parts of the landscape. Consequently, retention forestry practices can improve the provision of environmental regulation services (erosion resistance, regeneration ability of the plant community) which, in turn, may trigger the recovery of supporting (diversity and richness of plant species, greater stock of soil, and organic matter) and provision (timber, firewood, and forage) of ecosystem services.

Abbreviations

A: Annual; AU: Animal units; CV: Coefficient of variation; E: Exotic; N: Native; P: Perennial; SE: Standard error

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Authors' contributions

CAC and ML conceived and designed the experiment. LC, CAC, ML, and DRL performed field samplings. LC analysed the data and wrote the manuscript. All authors made substantial contributions in writing the manuscript. All authors read and approved the final manuscript.

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Availability of data and materials

The datasets used and/or analysed during the current study are available from the corresponding author on reasonable request.

Ethics approval and consent to participate

Not applicable.

Consent for publication

The authors consent to publish the data included in this draft.

Competing interests

The authors declare that they have no competing interests.

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