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A hard-to-keep promise: Vegetation use and aboveground carbon storage in silvopastures of the Dry Chaco

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ABSTRACT

In dry woodland regions, silvopastures have emerged as a promising option to balance cattle production, carbon storage and biodiversity. However, one of the major challenges in these systems, particularly when implemented in a matrix of natural vegetation, is the preservation of tree populations in the face of management actions implemented by ranchers to control woody encroachment. Here, we investigate the extent of that tradeoff by analyzing the impact of woody encroachment control practices on carbon storage in silvopastures of the Argentine Dry Chaco. First, we analyze tree density and carbon storage in aboveground woody biomass for silvopastures and woodlands at 24 sites in five properties across the Argentine Dry Chaco. Then, we characterize vegetation management goals and actions of ranchers who have adopted silvopastures in that same region, combining field assessments, high-resolution imagery analysis, characterization of site history, and surveys. We find that woody biomass in silvopastures retains an average of 64 % of the carbon present in aboveground biomass in intact woodlands (28.8 Mg C ha^{-1}). However, we also find that this storage capacity decreases by 12 % with each woody encroachment control intervention, due to these interventions' negative effects on tree density. Ranchers expressed concern about tree mortality, but also indicated low profitability of wood products and highlighted woody encroachment as a major issue for livestock production. Therefore, ranchers feel they have no choice but to continue preventing woody encroachment, even if this implies the gradual depletion of tree populations. Understanding how ranchers manage silvopastures, and how that management affects the provision of ecosystem services, is essential and will require more careful long-term monitoring and evaluation. This is particularly true in agricultural frontiers such as the Argentine Dry Chaco, where silvopastoral systems have the potential to mitigate the seemingly irremediable conflict between commodity production and nature conservation.

1. Introduction

Some of the greatest challenges in ecosystem management involve balancing the provision of multiple services that meet short-term societal needs with the long-term support of essential regulatory services (Chapin et al., 2009). Ecosystem management decisions can lead to win-win or trade-off relations between different ecosystem services, depending on the specific stakeholder demands that drive these changes (Ellis et al., 2019). Optimal management strategies consist in reconciling supply of, and demand for, ecosystem services. However, demand for ES varies widely among stakeholders (Lamarque et al., 2011), spatial scales (Sala et al., 2017) and geographical locations (Anadon et al., 2014). In rangelands, optimization can be particularly challenging because of the multiple demands on the system, including for example recreation for visitors in scenic landscapes, water provision for cities in urban watersheds, and fodder for ranching in productive areas (Yahdjian et al., 2015).

Besides food production, rangelands provide a wide spectrum of ecosystem services, including fiber production, carbon sequestration, sustaining biodiversity, and recreation (Sala and Paruelo, 1997). While

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only a few of these services have market value, rangeland managers face demands from multiple beneficiaries (Raudsepp-Hearne et al., 2010), particularly in semiarid and arid rangelands (i.e. grasslands, savannas, open woodlands, woodlands) where the vegetation is often comprised of a mix of herbaceous and woody vegetation in dynamic competition (Bond et al., 2003). For example, while aboveground net primary production can be higher when shrubs are kept (Eldridge et al., 2011), in areas where the prevailing land-use is cattle grazing, land managers in need of quality forage have an incentive to reduce woody plant cover as a means of maintaining or promoting livestock production (Archer and Predick, 2014). The challenge of combining food production with carbon storage and biodiversity protection in rangelands is even more salient in tropical and subtropical regions where the expansion of cattle ranching commonly drives deforestation and its associated carbon emission (e.g. Baumann et al., 2017; Nepstad et al., 2009).

Silvopastoral systems (SPS), in contrast to pure pasture systems, are better situated to balance multiple ecosystem services, while also producing more beef than woodlands. For that reason, SPS have been increasingly encouraged in the tropics as an alternative to the expansion of pure pasture over forests (Nair et al., 2009; Montagnini, 2017). SPS are characterized by the spatial arrangement of trees and pastures (e.g. lines of planted trees in pastures, or native trees randomly distributed within pastures), as well as the species composition and the types of management regimes used, which depend on the biophysical, economic, cultural, and market context of a region (Cubbage et al., 2012). Moreover, SPS can be implemented from different starting points (i.e. grasslands, degraded rangelands, native woodlands), each with differences in ecosystem management, as well as in economic and environmental performance (Jose et al., 2017). When SPS are implemented in native woodlands, their establishment implies the reduction of the shrub layers with different mechanical techniques such as ploughing (Daryanto and Eldridge, 2010) or roller chopping (Kunst et al., 2012).

Some aspects of SPS are well documented, be they negative (e.g., competition between grasses and trees or shrubs; Archer and Predick, 2014; Baldassini et al., 2018; Eldridge et al., 2011; Ford et al., 2019), or positive (e.g., benefits from livestock, such as favorable microclimate conditions for quality forage; Kallenbach et al., 2006; Karki and Goodman, 2015; Pang et al., 2019). Others, such as the medium- to long-term survival of trees and saplings and the evolution of carbon stocks in SPS, are less well known. While tree and sapling survival has been studied in ancient silvopastures such as the Dehesas and Montados of the Iberian Peninsula (Díaz, 2014; Moreno and Pulido, 2009; Pulido et al., 2010) or the rangelands of Australia (Saunders et al., 2003; Spooner et al., 2002; Manning et al., 2006), it remains poorly understood for regions where SPS are being implemented today and will likely expand in the near future. Although reports on carbon stock changes in SPS established on native vegetation exist across different regions (Aryal et al., 2018; McGroddy et al., 2015; Peri et al., 2017; Schneider et al., 2018) Somovilla Lumbreras et al., 2019), uncertainty remains around the impacts of some specific management actions (e.g., stocking rate adjustment and woody encroachment control) over woody vegetation structure and carbon storage.

Likewise, some studies have used surveys to examine various dimensions of SPS implementation, such as the determinants of SPS adoption (Calle et al., 2009) or discontinuance (Frey et al., 2012), ranchers' attitudes towards conservation (Plieninger and Modolell J. y Konold, 2004), or the perception and valuation by ranchers and other stakeholders of ecosystem services delivered by SPS (Garrido et al., 2017; Hartel et al., 2017; Surová et al., 2018). However, we are unaware of any research approaching ecosystem management in SPS from a socio-ecological perspective integrating rancher perceptions with field assessments of system dynamics. A better understanding of how land managers perceive particular ecosystem processes related to SPS management is key to designing more sustainable systems.

One of the regions where the area of SPS in native woodlands has been increasing the most drastically during the last decade is the Argentine Dry Chaco (Peri, 2016). This is due in large part to recent legal restrictions on woodland use and conversion embedded in the 2007 "forest law" (Law 26.331, Ministerio de Agricultura Ganadería y Pesca, 2015), which bans complete clearing of the tree canopy in much of Chaco region but allows partial clearing for silvopastures. Under the forest law, about 14 million hectares of woodlands are considered available for SPS implementation in the Argentine Dry Chaco. This constitutes an opportunity to expand food production and improve local livelihoods while sustaining multiple ecosystem services. In 2015, the National Institute of Agricultural Technology in collaboration with the Argentine Ministry of Agriculture and Fisheries and Argentine Ministry of Environment and Sustainable Developmen in Argentina agreed on a plan called Integrated Woodland and Livestock Management (MBGI in Spanish) (MAGyP et al., 2015), proposing a set of indicators to monitor SPS and improve environmental and production performance through adaptive management. However, at this time, there is no clear information about key ecosystem services such as carbon storage and their response to vegetation and cattle management in the Argentine Dry Chaco.

A better understanding of the impacts of silvopasture management on carbon storage and of the demand for, and perception of, silvopastoral systems by landowners is urgently needed for both the technical and political implementation of SPS. In this paper, we contribute to this understanding by evaluating carbon storage in trees and shrubs in SPS and comparing it with that of native woodlands for five fields situated in the Argentine Chaco. Furthermore, through the use of high-resolution images, we evaluate the response of tree density and carbon stocks to woody encroachment control events and other factors across multiple paddocks. Finally, we survey ranchers about their interest in keeping trees in cattle production systems and their opinion on the long-term viability of trees in SPS. In summary, we address the following three questions:

1 How much carbon is retained in aboveground woody vegetation in silvopastures of the Argentine Dry Chaco?
2 Are trees compromised under current vegetation management and if so, what are the implications for carbon storage?
3 How do ranchers perceive the benefits of keeping trees in SPS, and the viability of these trees in the long run?

2. Materials and methods

2.1. Study sites

The Dry Chaco (787.000 km2), the largest continuous expanse of dry forest on Earth (Portillo-Quintero and Sánchez-Azofeifa, 2010), has undergone rapid agricultural expansion during the last two decades (Baumann et al., 2016; Grau et al., 2005). With annual rainfalls concentrated in summer (December to March), ranging from 400 to 1000 mm/year from west to east (Sarmiento, 1972), soybean, maize and cattle ranching are the most common productive activities in the region, in the wetter and drier areas respectively (Houspanossian et al., 2016). Although it extends between Bolivia, Paraguay and Argentina, the largest part of the Dry Chaco (62 %) is located in the latter (Fig. 1). Within Argentina, the Dry Chaco represents close to 50 % of the country's native woodlands as well as the most active deforestation frontier (Gasparri and Grau, 2009; le Polain de Waroux et al., 2018). Woodland exploitation and cattle ranching have been the dominant land uses in the region since the beginning of last century (Morello and Saravia Toledo, 1959; Torrella and Adámoli, 2005). Currently, close to 23 % of the ranching area and 14 % of the cattle stock occur in woodland-dominated landscapes (i.e., landscapes with over 85 % woodlands) (Fernandez et al., 2020). This "woodland grazing" cattle production system includes both traditional systems based on natural vegetation, locally called "puestos" (Grau et al., 2008; Morello and

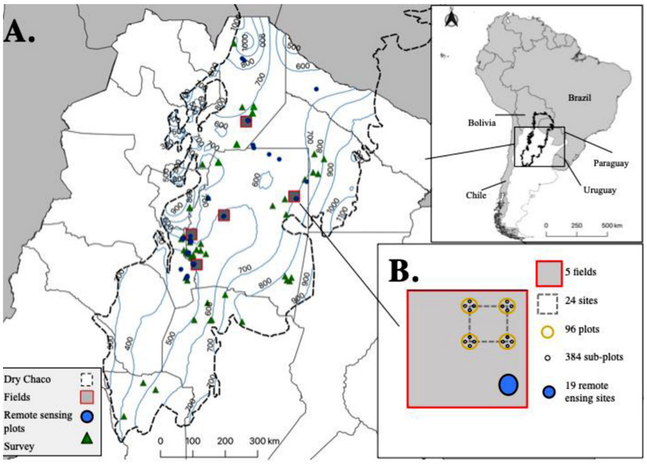

Fig. 1. Study area and sampling design. A = Study area with Dry Chaco location in South America in left and field (gray and red squares), remote sensing samples (blue circles) and surveys (green triangles) location in Argentine Dry Chaco on left. Blue lines represent isohyets of annual rainfall obtained through Hijmans et al. (2005). B = Carbon sampling and tree persistence (remote sensing sampling) design.

Saravia Toledo, 1959), and silvopastures in which part of the native cover is removed through the roller chopping technique (Fernandez et al., 2020). A 'roller-chopper' is an iron drum of 1.4 m diameter and 2.5 m width equipped with blades. It can be filled with 3000 kg of water, and it is usually pulled either by a small tractor, a four-wheel articulated tractor, or by a Caterpillar bulldozer (Peri et al., 2016).

2.2. Carbon storage estimations in SPS and woodlands

To determine the aboveground biomass stock in trees and shrubs, we sampled 24 sites in five cattle breeding/calving fields in the Argentine Dry Chaco between December 2016 and February 2018. In each of these five fields, one sampled site was a woodland, used as a baseline, and the other three to four sites were silvopastures. The main criterion for the selection of fields was that producers had a record of site management history. These fields were distributed across three provinces: one in Salta (district of Anta), three in Santiago del Estero (districts of Moreno, Lavalle and La Banda), and one in Catamarca (district of Fray Mamerto Esquiú) (Fig. 1 A).

Inside each of the 24 sites, we sampled tree and shrub biomass using a nested sampling design (Fig. 1B). At each site, we defined a 100 × 100 m square, and we placed one circular plot of 1000 m^2 (17.8 m radius) at each of the four vertices of that square, leading to a total of 96 sampling plots (see Fig. 1 B). First, we determined the aboveground biomass in trees. Inside each plot, we measured height and diameter at breast height (DBH) of all trees with a DBH greater than 10 cm (for a total of 1110 trees measured over the whole area), and identified the species of each of these trees. With these variables, we used an allometric biomass equation fitted for dry forests from Chave et al. (2005), determining tree biomass in kilograms. This method is a common approach to estimate aboveground biomass in the region (Gasparri et al., 2008).

In order to measure the aboveground biomass in shrubs, we sampled four sub-plots of 28.26 m^2 (3 m radius) inside each of the 96 sampling plots (Fig. 1 B). In these sub-plots, we measured the height and crown diameter of the largest axis and its perpendicular one for all shrubs over 30 cm of height and with less DBH than 10 cm. We measured a total of 3447 shrubs, identifying the species of each individual plant. We applied an allometric equation from Conti et al. (2019) to determine shrub aboveground biomass in kilograms (Supplementary equation S1). We defined the amount of carbon in aboveground biomass as 50 % of the estimated biomass (Gasparri et al., 2008). To estimate the average total carbon storage in aboveground woody biomass, three categories were considered separately: *woodlands* (five sites) as the references sites without roller chopping, *silvopasture with recent shrub control* (eleven sites), as the sites with a roller chopping and/or a shrub control in the last two years and *encroached silvopasture* (eight sites), as the sites with at least four years since the last disturbance. The ANOVA and Tukey test for normal variables, and Kruskal-Wallis and Wilcox-Mann-Whitney test, were performed to test significant differences between treatments (see Table 1).

2.3. Tree persistence and loss in SPS

To understand whether tree density in SPS decreased over time, we used a separate sampling design in which we randomly sampled 100 circular plots of 1 ha (hereafter "remote sensing samples") in 35 different paddocks spread over 19 fields across the region (Fig. 1 B; we sampled only those areas where high-resolution images were freely available in Google Earth). We used visual interpretation to count tree crowns inside each remote sensing sample using freely available images of Maxar's QuickBird satellite and aerial photos of Bing Maps in two different years, with the selection of year dependent on the availability of high-resolution images. In each remote sensing sample, we identified tree crowns in the oldest image, and then we assessed the persistence or loss of each tree crown with newer high-resolution images (Fig. 2 A). The period between consecutive high-resolution images, and thus between counts, varied from 2 to 15 years. We only counted tree crowns larger than 6 m in diameter (as measured in a Geographic Information System) in order to avoid counting shrubs, based on personal observations and knowledge of the area. This may have resulted in the omission of some small trees from the count.

Furthermore, we registered the year in which each paddock was transformed from woodland to silvopasture through visual interpretation of Landsat images in Google Earth (Supplementary figure S2). Landsat images, available in Google Earth from 1984, are sufficiently precise to allow the visual detection of transitions from woodlands to silvopastures through roller chopping. The years of SPS implementation range from 2001 to 2014. In order to detect whether the promulgation of the forest law in 2007 had effects on silvopastures structure, we

Table 1
Averages and standard deviations for carbon stock per hectare and number of individuals per hectare for trees larger than 10 cm of diameter at breast height (DBH) and shrubs larger than 30 cm in height in 3 categories: woodland, silvopasture with recent woody encroachment control and encroached silvopasture. Different superscript letters represents significant differences. Shrubs & saplings carbon and trees density are not normal variables for which comparisons was carried through Kruskal – Wallis test.

	Trees		Shrubs & Saplings		Total
	Carbon (Mg C ha^{-1})	Density (Ind ha^{-1})	Carbon (Mg C ha^{-1})	Density (Ind ha^{-1})	Carbon (Mg C ha^{-1})
Reference Woodlands (site = 5)	19.08 ± 7.34 [a]	225 ± 115 [b]	9.12 ± 6.52 [a]	4759 ± 2641[a]	28.21 ± 10.47[a]
Silvopasture with recent woody encroachment control (site = 11)	14.81 ± 7.70 [a]	86 ± 62 [a]	0.98 ± 1.00 [b]	2386 ± 1570[b]	15.80 ± 7.77[b]
Encroached silvopasture (site = 8)	17.18 ± 11.60 [a]	87 ± 81 [a]	3.15 ± 1.87 [c]	4758 ± 1893[a]	20.33 ± 11.85[a]

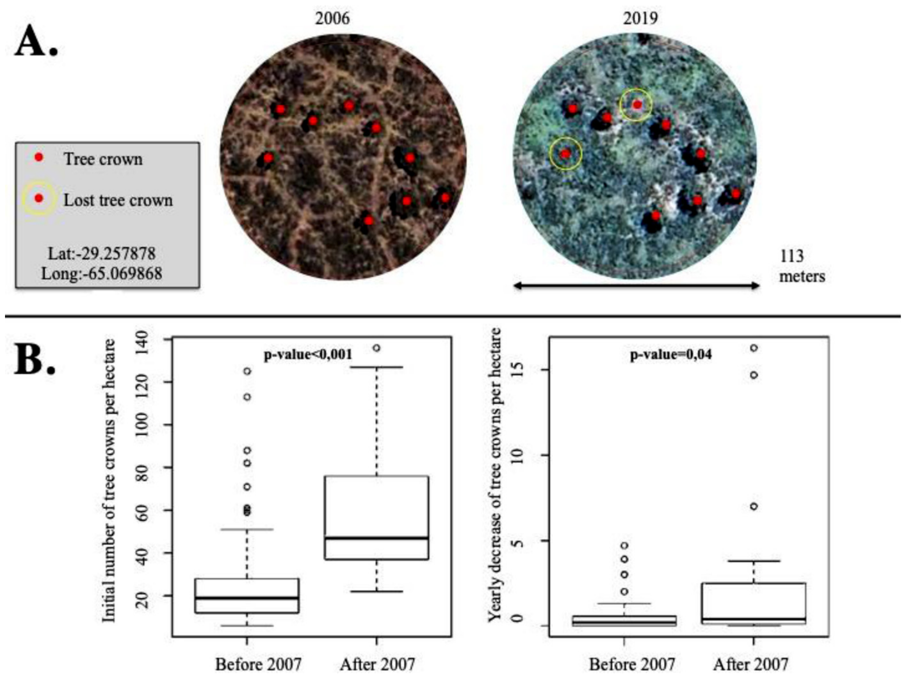

Fig. 2. Remote sensing sampling of 1 ha in two years (2006 and 2019). Red points are located at the center of crowns and red points with yellow circles represent tree crowns lost at the end of the period (A). The initial density and yearly decrease of tree crowns per hectare before and after 2007 are shown in (B).

classified remote sensing samples into two categories: those corresponding to pastures implemented before 2007, and those corresponding to pastures implemented after 2007. Finally, we calculated the average crown density for these two categories, as well as the average annual rate of crown loss for each remote sensing sample (Fig. 2). We used the non-parametric Wilcoxon-Mann-Whitney test in the R package 'coin' (Horton et al., 2012) for comparisons between categories (i.e., before and after 2007).

2.4. Tree density and carbon storage change in relation with vegetation management

The vegetation management history (understood as the number of woody encroachment control events, selective logging, or other disturbances), cannot be detected through the visual interpretation of satellite imagery, so it was reconstituted through historical management records collected in surveys with ranchers in the five fields were carbon assessments was made. We used generalized linear and linear mixed models (hereafter GLMM and LMM) to analyze the relationship between the independent variables, vegetation management related to woody encroachment controls, and dependent variables, carbon storage in tree biomass and tree density, for our set of 24 sites with aboveground biomass field assessments. Woodlands were included to represent situations with no disturbances. The independent variables were derived from the field management registers facilitated by landowners. These include: a) the number of woody encroachment control events; b) the type of woody encroachment control event used in each site, as an ordinal variable (i.e., roller chopping, roller chopping and either chemicals or fire control, or a combination of these three); c) time under SPS use (i.e., the number of years since first roller-chopping event); d) time since the last woody encroachment control event; and e) average annual rainfall at the site. We did not include any other environmental variable (e.g. annual temperature or evapotranspiration, which could be important in tree growth) because rainfall was the only variable measured by ranchers at each specific site, and no database of these variables exists at a sufficient spatial and temporal resolution.

In order to find out whether rainfall was a good descriptor of the environmental or management heterogeneity of our observational fieldwork, we applied a principal components analysis to all available variables at site level (n = 24). The first and second axes explain 62 % and 37 % of the variability respectively (Supplementary figure S3). The multivariate ordination showed that precipitation and number of trees explained a large part of variability of our sites, and that the remaining variables had less influence (Supplementary figure S3). This suggests that our unique environmental variable captures sufficient variability.

In total, we analyzed 95 plots for carbon storage (Mg C ha^{-1}) and tree density (individuals per ha) models in 24 sites (one plot was discarded as an anomaly: it was a patch of woodland inside a silvopasture paddock). The distribution of woody encroachment control events was: 20 plots of woodlands without any event (used as reference sites), and 75 plots of silvopastures with at least one event of roller chopping, of which 28 had one event (first roller chopping), 31 had two events, 13 had three events and 4 plots had five events. In order to deal with non-independence of the 1000 m^2 plots inside each site and control for spatial autocorrelation (Fig. 1 B), we used LMM and GLMM to fit tree density and carbon storage as response variables in two separate models. GLMM and LMM models avoid the problem of "pseudo-replication", as they structure the variance in two different types of effects: random and fixed effects (Zuur et al., 2009). In our case, random effects refer to the sampling design and are not of interest themselves for the hypothesis to be tested. However, these effects structure the variance-covariance matrix. We nested two levels of random effects to control for autocorrelation: one at field level and one at site level (Eq. 1). Fixed effects are the constants to be estimated from the data and relate to the ecological hypothesis (i.e., that trees are compromised under current vegetation management) (Eq. 1).

The two models to test the relation between explanatory variables and tree density and carbon storage takes the following general form:

$$Y_i = X_i\,\beta + Z_j + \varepsilon_i \tag{1}$$

Where Y is the dependent variable (i.e. tree density or carbon storage), the component $X_i\,\beta$ is the fixed effect term with X as the several explanatory variables (i.e. number of woody encroachment controls, etc. See 2.4 section), the component Z_j is the random effect (i.e., the 24 sites | 5 fields in the nested sampling structure), and ε_i is a vector of error terms. The use of random effects controls for the non-independence of plots at site level within each field.

The use of mixed models allowed retaining the variability between plots while discriminating the effects of the sampling design on the independent and dependent. Furthermore, the statistic distribution of the response variables (normal for carbon storage and negative binomial for tree density) was considered by analyzing error distributions. We used a variance function when data were heteroskedastic (Power variance for carbon stock; p = 0.0047). While an initial full model contained all explanatory variables, we identified appropriate minimum models based on inferential statistics (see statistics and p values in supplementary tables S4 and S5). Statistical analyses were performed in the software R (R Development Core Team, 2015).

2.5. Producers' perception of trees in SPS

Finally, we surveyed ranchers' perception of the value of keeping trees in cattle production systems and their opinion about the long-term viability of trees in SPS. To do this, the first author carried out 33 structured interviews with ranchers who implement SPS in six provinces of the Argentine Dry Chaco (Fig. 1 A). The interviewees were selected through purposive and snowball sampling (25 and eight ranchers, respectively). First, we interviewed the five ranchers who manage the sites where field measurements were made, as well as 15 additional ranchers who manage SPS, contacted through the National Institute of Agricultural Technology (INTA in Spanish). These ranchers then provided the contact information of eight additional ranchers. Random sampling in this context was not feasible due to the size of the study area. However, we were careful to cover a sufficient diversity of practices and profiles in order to have a representative sample of the people who implement silvopasture in the region (Fig. 1 A). The surveys asked ranchers to express their level of agreement with a series of statements about silvopastures. For this paper, we use a subset of ten of these questions that relate to tree management, the value of conserving them, problems with shrub controls, as well as barriers to the implementation and management of silvopastures.

3. Results

3.1. Carbon storage estimation in SPS and woodlands

The average carbon stored in the aboveground woody vegetation of woodlands was 28.21 Mg C ha^{-1}. SPS held an average of 64 % of that amount (56 % in sites with recent woody encroachment control event and 72 % in encroached sites) (p = 0.008). This difference was mainly due to the removal of shrubs and saplings in SPS, as the carbon stored in trees did not differ significantly (p = 0.43), storing on average 83 % of that stored in woodlands. Carbon storage in shrubs and saplings, by contrast, differed significantly between treatments (p = 0.001), and fluctuated between 11 % of that of woodlands for plots with recent woody encroachment control, and 34 % for plots in encroached silvopastures (p = 0.001) (Table 1).

Tree and shrub densities were more divergent between the SPS and woodlands than carbon storage. Tree density in SPS was 38 % of that found in woodlands (p = 0.01), whereas shrub density in SPS with recent woody encroachment control event was 50 % of that in woodlands (p = 0.001), but recovered 4–7 years after the woody encroachment control event (Table 1).

3.2. Tree persistence and loss in SPS

3.2.1. Remote sensing samples

Of all the remote sensing samples measured, 77 % showed a decrease in the density of tree crowns between two different periods (Fig. 3 a). The yearly reduction rate of crowns larger than 6 m of diameter ranged between and average of 0.47 crowns per year, or one crown lost every 2.13 year (for samples with SPS implemented before 2007), to an average of 0.83 crowns per year, or one crown lost every 1.2 years (for those with SPS implemented after 2007) (p = 0.04) (Fig. 2 b). Noticeably, the initial tree crown density was higher on average in SPS implemented after 2007, than in SPS implemented before 2007 ($p < 0.001$) (Fig. 2 b). The remote sensing samples with a high initial number of tree crowns commonly presented an accelerated rate of decrease (Fig. 3).

3.2.2. Effect of management on tree density and carbon stocks

The implementation of repeated woody encroachment controls (including the first roller chopping to convert woodlands to silvopastures) resulted in a linear reduction of carbon storage in trees of 12 % per event (Fig. 4 A, Equation 2). Tree density, on the other hand, decreased at a negative exponential rate, with a major reduction after the first disturbance. These results suggest a decrease in trees (and in carbon stored), not only at the time of implementation of silvopastures, but also throughout its existence, due to subsequent woody encroachment control events (Fig. 4 B, Equation 3).

Carbon stock in trees (Mg C ha-1) = 19.18(± 2.23) - 2.25(± 0.72)* Number of woody encroachment controls (2)

Trees (# individuals ha-1) = EXP 5.15(± 0.19) - (0.42(± 0.09)*Number of woody encroachment controls) (3)

In the mixed model regressions, only the number of woody encroachment controls was significant in both the carbon storage model (p = 0.0029) and tree density model (p = 0.0001). The variables 'number of woody encroachment control events', 'different types of woody encroachment control event used in sites', 'length of the paddock's use as an SPS', 'time since the last woody encroachment control event occurred' and 'average annual rainfall' were not significant (Supplementary tables S4 and S5). The variance explained by fixed effects in the carbon storage model was 0.26 and the variance explained by the complete model (fixed effects plus random effects) was 0.97. The pseudo R^2 for the tree density model was 0.64, with a variance explained by fixed effects of 0.33.

3.3. Producer perception of trees in SPS

The majority of ranchers we interviewed agreed that SPS improved animal welfare (85 % of respondents said they either moderately or strongly agreed with the statement) and declared not to perceive a decrease in grass productivity with higher tree coverage (61 %; Fig. 5). In addition, a majority agreed that SPS are more visually pleasing than pure pastures (88 %) and that they are better for biodiversity conservation (85 %). These opinions about SPS could constitute the main reason for the shared concern about trees mortality (65 %) and widespread intentions to conserve trees in SPS (82 %). However, 70 % of the producers also declared that wood products did not represent a relevant income for their farms, which are mainly sustained by cattle production. Moreover, the majority agreed with the statement that woody encroachment was the main technical challenge for cattle ranching (88 %), and that shrubs reduced cattle stocking rate capacity (85 %). Finally, there was no clear consensus in the interviews about a positive relation between woody encroachment rate and tree density (Fig. 5).

4. Discussion

4.1. Carbon storage in SPS

Silvopastures in native woodlands in the Dry Chaco represent a promising way to achieve carbon storage while also producing high quality food. However, evidence of the impacts of current SPS management practices on ecosystem services, as well as on the perception land managers have of these impacts, is still very scarce. Here, we combined field and remote sensing assessments, regression analyses

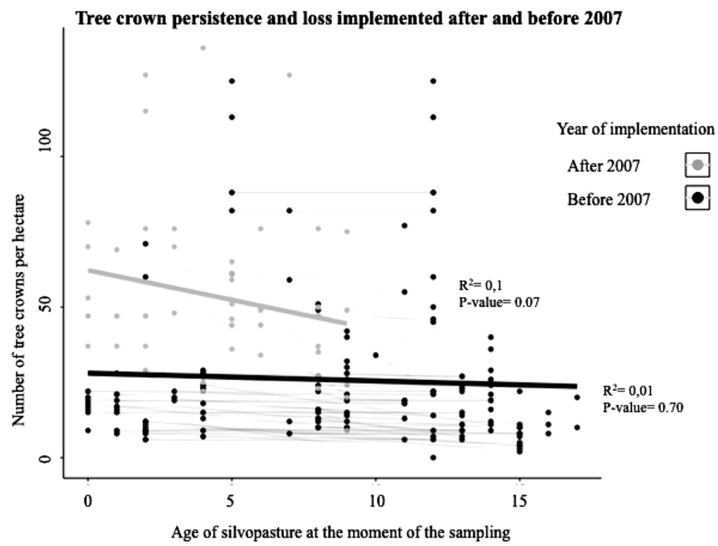

Fig. 3. Tree density in relation with the age of the silvopasture at sampling time. In gray the series after 2007 and in black the series before 2007.

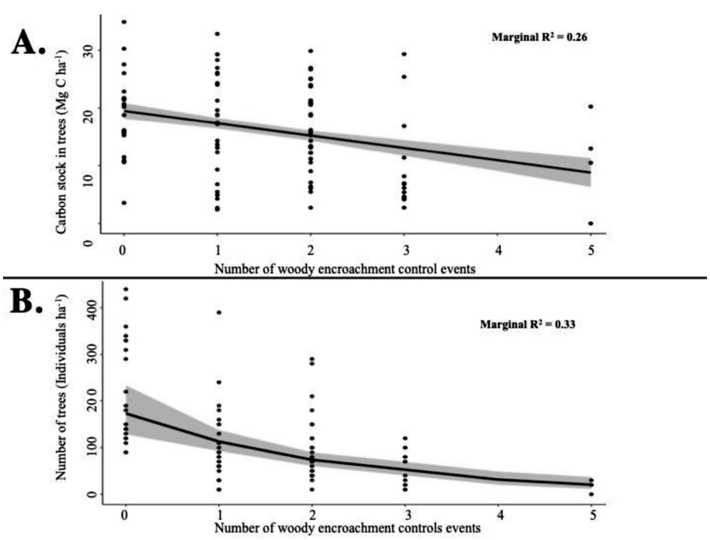

Fig. 4. **A** = Relationship between the number of woody encroachment control events and carbon stock (Mg C ha^{-1}). **B** = Relationship between the number of woody encroachment control events and tree density (Individuals ha^{-1}). In both cases, undisturbed woodlands were included as the as zero disturbance level. Marginal R^2 represent the variance explained by fixed effects in the Generalized linear mixed models.

and surveys to elucidate the role of SPS for carbon storage and its dynamics under current management practices in the Argentine Dry Chaco. Our findings suggest that despite an important reduction in tree density and shrub biomass with roller chopping, close to 70 % of the aboveground woody biomass of woodlands can be held in silvopastures. However, we found that current woody encroachment control practices can reduce carbon stock and tree density, leading to a progressive deterioration of the co-production of carbon and food production. Finally, while farmers identified several benefits of silvopastures and voiced concern over tree loss, they remarked that woody encroachment represents a drain on farm income, which suggests that improvements in technology and management are urgently needed in order to sustain the provision of multiple ecosystem services in this deforestation frontier.

SPS should be considered as a potentially important supplier of ecosystem services in terms of carbon storage in the Dry Chaco. Our analysis reveals that carbon storage in aboveground woody biomass in SPS is on average 64 % of that stored in woodlands. This is the case despite a drastic modification in shrub cover (a 70%–90% decrease) because the majority of carbon storage is located in large trees (Table 1). If exotic pastures are planted during SPS implementation, rather than keeping native grasses, they can produce between 4000 and 12,000 kg of dry matter forage (Kunst et al., 2012), which represents 4–12 times more than the herbaceous forage produced in woodlands of the region (Kunst et al., 2012; Morello and Saravia Toledo, 1959; Rueda et al., 2013). In addition, SPS in the Dry Chaco can store on average approximately five more megagrams of soil organic carbon (SOC) per hectare (+13 %) than intact woodlands in first the 40 cm of depth (Somovilla Lumbreras et al., 2019), which could offset the carbon reduction on aboveground biomass of roller chopping. This implies that very large increases in food productivity can be achieved with moderate tradeoffs in terms of carbon storage.

We warn against extrapolating these ecosystem service values to the entire Dry Chaco. To retain a high fraction of the carbon stored in woodlands, it is necessary for SPS to maintain a structure consisting of matures trees that normally compose SPS (i.e. *Aspidosperma, Prosopis, Schinopis, Zyziphus* species genera. In woodlands with a previous history of degradation or with edaphoclimatic limitations (salinity or regular flooding), the total carbon stock is commonly lower, with a higher fraction represented by the shrub layer (Kunst et al., 2006). In such situations, the implementation of roller chopping could lead to lower carbon stocks in SPS. Peri et al. (2017) simulated different intensities of SPS implementation in 50 woodland locations covering different conditions in terms of structure and carbon pools in the dry Chaco, which had been measured by Gasparri and Baldi (2013). They showed that in some cases, the initial conditions of the woodlands are not sufficient for the implementation of desirable SPS (for example in terms of large tree

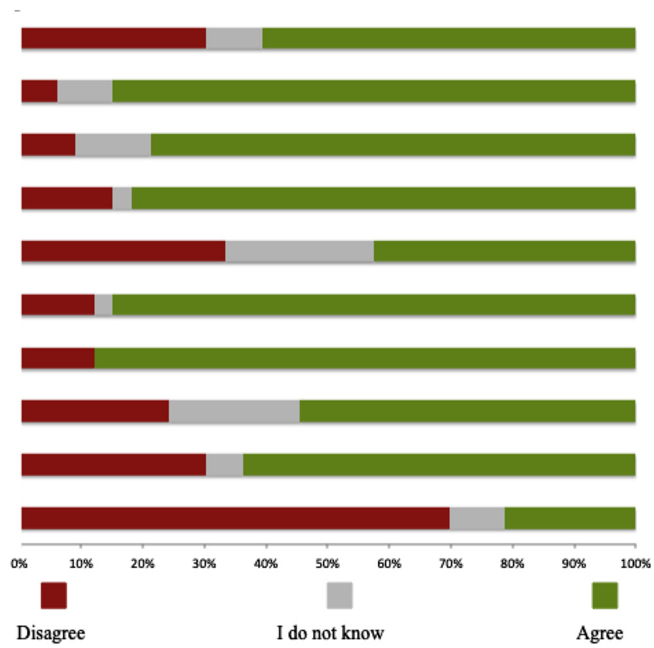

Fig. 5. Percentage of agreement or disagreement of ranchers who implement silvopastoral systems in questions related to tree benefits, tree mortality and woody encroachment management.

density). While our current work shows that SPS can retain between 56 % and 72 % of the carbon stored in woodlands, in the tropics, this values falls to 29 % for secondary forests and 7% for primary forests (Kauffman et al., 2009; McGroddy et al., 2015). In an assessment of carbon storage carried out on the Southern border of the Dry Chaco, Peri et al. (2017) also found that SPS retained either 46 % or 24 % of aboveground carbon storage in comparison to woodlands for SPS with and without shrubs respectively.

Lack of knowledge of how the carbon sequestration rate of different carbon sinks varies across management types is a major gap for SPS in dry forests worldwide. Studies of the dynamics of woody carbon in secondary dry forests in Mexico (Aryal et al., 2014) found a range of carbon sequestration rates from 2.2–4.7 Mg C ha^{-1} with a forest age of 20 years and four years, respectively. However, information of tree or shrub growth, under roller chopped silvopastures is to our best knowledge nonexistent. We believe that the re-measurement of these SPS permanent plots will help us improve the understanding of ecological functioning and temporal dynamics, in a system where the rate of carbon sequestration may differ from a complete logging due mainly to the age of the standing trees. This information will be valuable for programs such as REDD+ (Rosenstock et al., 2019) as well as for validating carbon balance models.

4.2. SPS management and decrease in tree density

Tree density decreased in the majority of our sampled paddocks. Rather than the result of an intention by farmers to get rid of trees, our interviews indicate that this could be collateral damage from woody encroachment controls. This shows, as pointed out in other studies and reviews of SPS (Jose et al., 2017; McGroody et al., 2015; Peri et al., 2017) and in ecosystem services evaluations (Núñez-Regueiro et al., 2019), the importance of taking into account management practices in the medium- to long-term supply of ecosystem services. In our case, while tree density decreased in the majority of our remote sensing samples, decrease was higher in denser plots (Fig. 3). This could be due to the fact that having less open space for machinery operations increases the risk of damage to trees, or from decisions by producers to leave fewer trees in order to facilitate cattle management. This trade-off between pasture management and carbon storage, also manifest in the negative relationship between the number of woody encroachment controls and tree density and carbon storage, represents a challenge for the long-term provision of carbon storage in these systems. Woody encroachment is a prominent issue perceived as serious threat to incomes from cattle production.

The effect of encroachment control on carbon storage was lower than its effect on tree density, probably due to the low intervention in basal area (the area of land that is occupied by the cross-section of tree trunks and stems at the base) at the time of initial roller chopping (between -4% to -9% of woodland basal area) (Navall, 2012). Basal area is the main determinant of woody biomass and thus of carbon storage (Chave et al., 2005). Tree growth could increase after disturbances that release water shrub's competition, which could contribute to increasing carbon sequestration years after SPS implementation. Surprisingly, no significant relationship was found between carbon storage and tree density for the years since SPS was implemented. This lack of correlation may be attributed to spatial heterogeneity or possibly to the fact that sites analyzed are very recent (from 0 to 15 years).

We found one prior report on tree mortality and damage from roller chopping (Kunst et al., 2015), in which tree density decreased by 14 % after one roller chopping, consistent with our estimates. Furthermore, Kunst et al. (2015) assessed trees smaller than 1.3 m in height, showing that enough regeneration existed to sustain diverse tree diameter classes. Studies of tree decrease in woody-herbaceous ecosystems in multiple regions have suggested a degradation process caused by a general lack of tree management, particularly in the recruitment of new individuals (Manning et al., 2006). We found shrub and sapling densities to be similar, but we also found a similar proportion of sapling and shrubs across all years, reflecting non-selective management across years after the last disturbance (Supplementary figure S6). This suggest that recruitment exist despite grazing pressure which means that tree degradation processes could be reverted with active practices in vegetation management, for example marking saplings (to be identifiable from a tractor) before a roller chopping. Studies assessing tree recruitment in SPS have been carried out particularly in Dehesas and Montados. These studies found that the probability of securing tree regeneration was low and suggested implementing the enclosure of paddocks and exclusion of livestock for long periods (Pulido et al., 2010; Ramírez and Díaz, 2002). However, *Quercus ilex* and *Quercus suber* are palatable for livestock, which could be a significant difference with the Dry Chaco's forest dominated by trees with thorns and coriaceous leaves. For the long term sustainability of SPS in Chaco, a better understanding of tree regeneration under grazing, encroachment and their management is urgently needed.

A new SPS design called Integrated Woodland and Livestock Management (Manejo de Bosques y Ganadería Integrado or MBGI in Spanish) was recently. This operational framework is based on adaptive management, and has the objective of developing a system that promotes tree management, securing different diameter classes over time (MAGyP et al., 2015). This outline proposes cycles with temporary exclusion of cattle in order to regenerate minor trees and restore SPS structure, similar to the experience of Dehesas. In light of our results suggesting a lower economic importance of wood products in farm systems, we suggest that it is also urgent to develop forest-based value chains in order to integrate woodland management in entrepreneurial decisions (e.g. stimulating charcoal or fire wood, or fence post production), as well as to avoid undue opportunity costs of closure of paddocks to cattle.

A number of sources of uncertainty in our analysis are worth mentioning. First, this study is focused on medium- to large-scale producers, and it is likely that smaller producers have different perceptions of woodland products, as well as different management techniques (Cáceres et al., 2015). Further work should attempt to understand perception and management impacts across different social actors of Dry Chaco. Second, climatic disturbances such as drought, strong winds or hail were not considered, but they could have effects on woodland carbon balances (Brando et al., 2019) and likely on tree mortality. Third, in the time period of the study (2016–2019), only four of the 24 sites experienced a new woody encroachment control event. Two of this sites presented tree losses after the events and two did not. Interestingly, the last two were sites with low tree density. We hope to remeasure tree density and biomass in our sites. We suggest that it will be necessary to increase the number of permanent SPS and woodlands plots to continue analyzing the impacts of SPS management in ecosystem services. Finally, a recent meta-analysis of woody plant removal indicates that an ecosystem response to woody plant removal is driven by removal methods (Ding et al., 2019). Despite the evidence presented in this work about the effects on carbon storage, we did not go into any detail about the differentiated impacts of alternative woody encroachment control techniques. In order to measure those, it would be necessary to conduct experiments with fire, chemical controls and the more classic roller chopping, as well as with different stocking rates.

5. Conclusions

In this study we showed that silvopastures hold promise for reconciling food production with important ecosystem services in dry woodlands. However, this promise is made more precarious by management tradeoffs faced by ranchers. We found that silvopastures hold 63 % of the total carbon stored as aboveground woody biomass in woodlands, which suggests that SPS are a good supplier of this ecosystem service, mainly through the retention of large trees that represent the largest carbon pool. Moreover, the national forest law appears to have affected SPS structure, with higher tree densities after its promulgation. However, both tree density and carbon storage decreased over the years, with a higher decrease in SPS with high density of trees at the onset. The main reason for this phenomenon was recurrent woody encroachment controls, which suggests a trade-off between woody encroachment and tree survival. In addition, while the majority of producers that we interviewed expressed concern about tree loss, they also said that wood products do not represent an important source of income in their production system and that woody encroachment represents a major nuisance. This highlights the prioritization by managers of increasing and maintaining pasture productivity over managing tree populations.

In the near future, SPS may be implemented on as many as 14 million hectares of woodlands in the Argentine Dry Chaco, one of the most active agricultural expansion frontiers worldwide. Well-managed SPS have the potential to mitigate the environmental tradeoffs associated with increased food production in this region. Poorly managed SPS, on the other hand, may be but a step towards complete deforestation. In order to prevent undue degradation of natural resources in the region and optimize SPS as a way to increase production with fewer environmental impacts, it is imperative to develop formal systems for monitoring tree cover in SPS and improve practices to control woody encroachment that minimize damage to large trees. Along with the development of value chains for forest products and their incorporation into ranchers' economic portfolios, these actions may help steer SPS in the right direction and give them a chance to deliver on their promises.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.agee.2020.107117.

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