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Comparison of animal biodiversity in three livestock systems of open environments of the semi-arid Chaco of Argentina

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Abstract. Increasing global food demand requires the exploration of agricultural production systems that minimise the conflict between food production and biodiversity conservation. Cattle ranching is a main land-use in tropical and sub-tropical South American semi-arid ecosystems, such as the Chaco eco-region of sub-tropical Argentina, one of the most active frontiers of land-use change. Despite open habits being a key component of the Chaco landscape, conservation studies and policies have focussed on forests. In this study, bird and mammal communities of three different open-canopy livestock-producing systems in the semi-arid Argentinian Chaco: natural grasslands, sown non-native pastures and silvopastoral systems are discussed. Diversity (Inverse Simpson index) and species composition (multivariate ordinations) were measured and species identified that characterise each system (indicator species). The three livestock systems did not significantly differ in terms of diversity but showed differences in the composition of bird communities. Natural grasslands had the highest number of bird and mammal indicator species (including *Myrmecophaga tridactyla*, a high conservation-value species). These results highlight natural grasslands as a landscape unit with a high conservation value and indicate that they should be explicitly targeted by conservation and land-use policies, particularly because they represent a small and rapidly decreasing proportion of the semi-arid Argentinian Chaco.

Additional keywords: biodiversity conservation, native grasslands, rangeland communities, savanna ecology, semi-arid rangelands.

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Introduction

As global food demand increases, conserving biological diversity becomes a greater challenge (McAlpine *et al*. 2009). This situation has encouraged the scientific community to explore in detail the conflicts between agriculture production and biodiversity conservation, and the ways to mitigate it (Chará and Giraldo 2011; Murgueitio *et al*. 2011). Understanding the interaction between human pressure and response factors in ecosystems is a key issue for the implementation of management policies, when aimed at minimising the effects of land-use on biodiversity (Mattison and Norris 2005; Fischer *et al*. 2008). Although this relationship has been the subject of much theoretical debate (for example, Grau *et al*. 2013), it has not been fully evaluated in empirical terms; which makes this area of research a priority for agronomic (Pretty *et al*. 2010) and conservation (Sutherland *et al*. 2009) purposes.

The Chaco eco-region of dry forest is one of the most active frontiers of land-use change in South America (Aide *et al*. 2013; Graesser *et al*. 2015). Given the high suitability of the area for agricultural production, the conflicts with biodiversity conservation are increasing (Lambin *et al*. 2013). There are ongoing regional initiatives to mitigate this conflict (for example, Law of Minimum Standards for Environmental Protection of Native Forests –26331-; REDD+) but they tend to neglect the importance of natural grasslands as a conservation target, partly based on the assumption that biodiversity co-varies with biomass and that both variables are maximised in well developed forests (Phelps *et al*. 2012; Strassburg *et al*. 2012). In landscapes historically shaped as a mosaic of forest and open areas, however, a substantial part of the biodiversity depends on open landscapes for survival and this appears to be the case of the dry Chaco (Macchi *et al*. 2013; Torres *et al*. 2014; Grau *et al*. 2015).

In this paper, we consider natural grasslands, sown pastures and silvopastoral systems, as livestock systems of productive open habitats of the semi-arid Argentinian Chaco. The main difference between the sown pastures and silvopastoral systems is the presence of native tree species in the latter. Grasslands currently represent ~7% of the semi-arid Argentinian Chaco (satellite images of 2007; Gasparri and Baldi 2013). Although the areas of natural grasslands are rapidly shrinking, being replaced either by sown pastures or by soyabean crops (Grau *et al*. 2015), systems based on sown grasses are expanding. A potentially key factor in the direction of land-use changes is land tenure (Demsetz 1967; Hardin 1968; Beaumont and Walker 1996; Altrichter and Basurto 2008) for both economic and cultural reasons.

Natural grasslands in the semi-arid Chaco can be found in protected areas, indigenous properties and *puestos* systems (poor peasant livestock operations) as management of natural resources under current land tenure does not involve the conversion of their original land cover into others. Instead, sown pastures and silvopastoral systems (*desbajerado*) are anthropogenic systems of large private properties with livestock production, requiring a high level of land conversion associated with a large economic and technological investment.

As these three livestock systems are difficult to discriminate in satellite imagery (Clark *et al*. 2010), they are often included in a single category in land-use planning schemes (for example, Argentine Forest Law). However, they differ in terms of their ecological functions, environmental services (Laclau 2012) and biological communities (Macchi *et al.* 2013). The presence of trees in silvopastoral systems, for example, offers economic and environmental benefits, including soil enrichment, carbon storage, shelter for wildlife, and a shady microenvironment that reduces heat stress for livestock and pastures (Jose 2012). However, natural grasslands provide habitat for numerous vertebrate species adapted to this system (Macchi *et al*. 2013; Torres *et al*. 2014).

In this paper we focus on the three livestock systems of open environments of semi-arid Argentinian Chaco; and we compare them quantitatively and qualitatively in terms of their bird and mammal communities. We hypothesise that the management of the different open areas affect biological communities in different ways. A dominance of native grass species over exotic grass species could be a key factor for biological communities so we should expect to find the highest diversity of birds and mammals in natural grasslands (prediction 1). The presence of trees is also a requirement for many species of birds and mammals and, therefore, we could alternatively expect silvopastoral systems to be the most biodiverse habitat (prediction 2). Any difference revealed among livestock systems in terms of biodiversity should be also stated in terms of the composition of plant communities. If the dominance of native grasses overrides the presence of trees, natural grasslands should shelter more distinctive communities compared with sown pastures and silvopastoral systems (for example, segregated in ordination diagrams, prediction 3). However, if the presence of trees is a more important factor for biological communities, then silvopastoral systems should be the most segregated treatment in the ordination space (prediction 4).

To assess these predictions, we pursued three specific objectives: (1) to compare the diversity (Inverse Simpson index) of birds and mammals in three livestock systems in open environments (natural grasslands, sown pastures and silvopastoral systems); (2) to assess the patterns of similarity in community composition among the management systems (using non-metric multi-dimensional systems based on the relative frequency of species); and (3) to identify particular species that characterise each land-use system (through identification of indicator species).

Methods

Study area

The study was conducted in the northern Argentinian sector of the dry Chaco eco-region (Fig. 1). Sites of natural grasslands, sown pastures and silvopastoral systems were located in eastern Salta province, western Formosa and Chaco provinces and the northern sector of Santiago del Estero province, spreading over a total area of 172 800 km^2. Natural grasslands are mostly in indigenous properties and protected areas. Both land tenures are much less widespread than large properties in the region, which include extensive sown pastures and silvopastoral systems. As a consequence of this, we included a lower number of sites of natural grasslands than for the other two types, reflecting the situation in the dry Chaco.

There are two basic types of natural pastures: *Aibales* are natural grasslands dominated by *Elionurus muticus* (‘*aibe*’), which occur on abandoned river beds with sandy, well drained soils; and *Espartillales*, which develop on periodically burnt areas associated with the north–south oriented desiccant winds, and are dominated by bunch grasses belonging to *Trichloris*, *Gouinia* and *Setaria* genera (Morello *et al*. 2009). *Aibales* are the most common grassland type in the northern portion of the Argentine Chaco, where this study was focussed. To obtain comparable samples we only included *aibales* in this work.

The introduction of domestic livestock in the past century led to the removal of the fine fuels and favoured woody encroachment over grasslands (Adámoli *et al*. 1990). In protected areas, grasslands are declining as a result of fire-suppression practices that favour the replacement of native grasses by woody species (for example, Copo National Park in Santiago del Estero province; Cardozo *et al*. 2011; Grau *et al*. 2015). When grasslands are outside protected areas, fire is usually set to induce regrowth of a higher nutritive value, which is preferred by livestock (Morello *et al*. 2008). This dynamic prevents woody invasion and promotes the regrowth of native grasses (Casillo *et al*. 2012), increasing the diversity of herbaceous species (Kunst *et al*. 2003). In large private properties, natural grasslands are often replaced by soyabean and other annual mechanised crops, and by sown pastures (Grau *et al*. 2015). However, there are still large areas of natural grasslands, mainly in indigenous communities, private areas and *puestos* in Santiago del Estero province.

Sown pasture systems involve the replacement of natural grasslands or the complete removal of woody vegetation before sowing exotic species of grasses, to support livestock grazing. The most commonly used species in semi-arid Chaco are the African bunchgrasses *Panicum maximum* (cv. Gatton panic) and *Cenchrus ciliaris* (cv. Buffel grass). In these systems, the high stocking rate of livestock leaves little chance of shelter and food for native fauna. Due the high degree of economic investment involved, these systems are only found in large private properties.

Silvopastoral systems combine livestock, trees and grasses in the same management unit, sometimes adapting forestry practices to livestock production. In these systems, the woody understorey is mechanically removed, allowing grasses to grow under a relatively open tree canopy. Unlike the full conversion of forests into sown pastures, which involves total deforestation, forest conversion into silvopastoral systems is allowed in the ‘yellow areas’ (extensive management) of the Law on Minimum Standards for Environmental Protection of Native Forests (Quispe Merovich and Lottici 2011). However, in silvopastoral systems of semi-arid Chaco, the most commonly used grass species are the same used in sown pasture systems; and

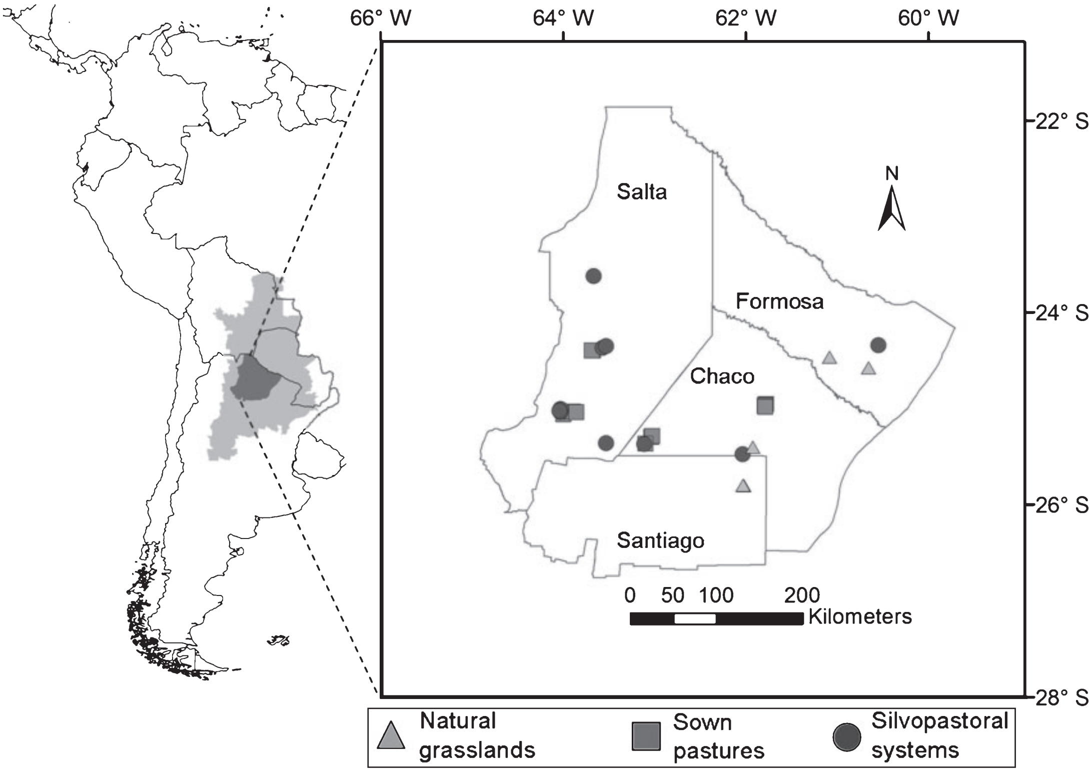

Fig. 1. Location of the study area in South America. Gran Chaco is depicted in light grey and study area in dark grey. The enlargement of the study area shows the location of sampling sites.

silvopastoral systems tend to gradually convert into sown pastures after the tree layer deteriorates due to the reduced regeneration capacity of tree species in a heavily grazed and frequently burnt environment. Compared with sown pastures, silvopastoral systems have higher soil fertility, carbon stocks, and a shady environment, which maintains the nutritive value of grasses into the dry season and shelters livestock, and thus reducing heat stress (Jose 2012). However, trees do compete with grasses for soil, water and light, which results in a trade-off between herbage production for livestock and canopy cover (Sharrow 1999). This type of land-use is also typically found on large private properties.

Study design

We conducted bird surveys at 30 sites distributed among natural grasslands ($n = 5$), sown pastures ($n = 11$), and silvopastoral systems ($n = 14$). In 26 of those 30 sampling sites, we also sampled mammals (respectively 5, 10 and 11 sites) because in three sites of silvopastoral systems and one of sown pastures, the height of the grasses did not allow us to conduct the sampling protocol. Sampling was conducted during the dry season (May–September) of 2009, 2010 and 2011, at peak hour of activity of birds (0800–1030 hours; Ralph *et al*. 1996).

Each site consisted of a 900 by 900-m area with nine points and nine transects evenly distributed (i.e. 300 m of distance between points and a buffer area of 150 m around the external points of the area). At each sampling point we recorded every terrestrial bird, seen or heard within a span of 10 min and a radius of 30 m. Between two points, we set a 100 by 4-m transect, where we recorded every trace of terrestrial large- and medium-sized mammals (sightings of living individuals, tracks or faeces). We only considered species with an average weight larger than 1 kg, because their tracks and faeces were visible to the human eye (i.e. we excluded small rodents and small marsupials). For the analysis, we only included species that were present in at least three sites in total (this criterion left out *Didelphis albiventris*). In cases when tracks were too similar between two or more species as to allow them to be reliably discriminated, we created groups of species: ‘Canidae’ (*Cerdocyon thous* and *Lycalopex gymnocercus*), ‘Small Dasypodiadae’ (*Cabassous chacoensis*, *Chaetophractus vellerosus*, *Chaetophractus villosus*, *Dasypus septemcinctus* and *Tolypeutes matacus*), ‘Small Felidae’ (*Puma yaguarondi* and *Oncifelis geoffroyi*), ‘Tayassuidae’ (*Catagonus wagneri*, *Pecari tajacu* and *Tayassu pecari*) and ‘Rabbits’ group; this last one includes a real rabbit (*Sylvilagus brasiliensis*, Fam. Leporidae, Orden Lagomorpha) and the chacoan cavy, a rodent called ‘conejo de los palos’ (i.e. ‘rabbit from the woods’) by local people (*Pediolagus salinicola*, Fam. Caviidae, Orden Rodentia) as both species are ecologically and morphologically similar. This classification left the following single species (i.e. not included in groups): *Euphractus sexcinctus*, *Mazama gouazoubira*, *Myrmecophaga tridactyla*, *Procyon cancrivorus* and *Puma concolor.*

Data analyses

To quantitatively analyse differences in biodiversity among the birds and mammal communities of livestock systems in open environments (first research objective), we calculated the Inverse Simpson index (C_{inv}; Feinsinger 2003) as:

$$C_{inv} = \frac{1}{\sum_{i=1}^{s}(p_i^2)}$$

where $p_i = SF_{ik}/N_k$; SF_{ik} is the frequency of i species in site k (a given species in a given site); and N is the sum of frequencies of every species in site k. Each species frequency (SF) at a site was calculated as the number of sampling points where such bird species was registered, or of sampling transects where a mammal species (or group of species) was recorded. The values ranged from 0 (when the species was not recorded) to 9 (when it was found at each sampling point or transect). C_{inv} has a minimum value of 0 (for samples with no species) and a maximum value that equals the total number of species when all species have exactly the same frequency in the site. C_{inv} is more sensitive to changes in equality of species than, for example, the Shannon–Weaver index (Feinsinger 2003). We then performed an ANOVA and Tukey comparisons among livestock systems to assess differences in C_{inv}. We checked for normality and homogeneity of variance by using standard graphical methods (Di Rienzo *et al.* 2011). To assess if the potential differences among groups were associated with their geographic distribution, we conducted correlograms with Moran index (Legendre and Legendre 1998) using eight classes with equal number of pairs of compared points, and it showed no spatial structure in C_{inv} values of any biological group ($P > 0.05$).

To evaluate how the composition of the communities of birds and mammals differ among livestock systems (second research objective), we used non-metric multi-dimensional scaling ordinations (Kruskal and Wish 1978) based on a matrix of Bray–Curtis distances (Legendre and Legendre 1998) between pairs of sites. In computing similarity between two sites, the Bray–Curtis index compares the frequency of each species present. Sites sharing species with comparable frequencies are considered more similar, and are thus closer in the multi-variate space (Jogman *et al.* 1995). In contrast to other ordination techniques, non-metric multi-dimensional scaling has no assumptions about how species are distributed along underlying environmental gradients (Kenkel and Orlóci 1986). We performed autopilot procedures (‘slow and thorough’ mode) to decide the best solution (McCune and Grace 2002). To generate acceptable ordinations, we used three-dimensional configuration for bird ordinations, because final stress (an index of agreement between the distances in the graph configuration and the distances in the original data matrix) was 13.98 (most ecological community datasets have solutions with stress between 10 and 20, and less than 20 is usually considered acceptable), and was different from chance (Monte Carlo: 500 runs with randomised matrix, $P = 0.004$; McCune and Grace 2002).

Mammal frequency data showed a weaker structure, and non-metric multi-dimensional scaling did not converge into a configuration with acceptable stress values (<20) in three or less ordination axes. We, therefore, conducted mammal ordinations by means of the Bray–Curtis ordination procedure (Bray and Curtis 1957), based on a matrix of Bray–Curtis distances between pairs of sites (Legendre and Legendre 1998). Bray–Curtis ordination extracts ecological gradients with no assumption of a linear relationship between species (McCune and Grace 2002). To arrange points along the diagram, Bray–Curtis ordination selects reference points and uses a distance matrix to locate the remaining points in a relative way. We used the variance-regression method (Beals 1984). The first end-point has the highest variances of distance to other points. This finds a point at the long end of the main cluster in species space, and it ignores outliers because their distances will be consistently higher, so resulting in a lower variance (McCune and Grace 2002). We based non-metric multi-dimensional scaling and the Bray–Curtis ordination on frequency of species with presence in at least three sites. In both ordinations, we calculated the proportion of variance represented by ordination axes at the final configurations. For non-metric multi-dimensional scaling, we calculated the coefficient of determination (r^2) between distances in the ordination space and distances in the original space, by the ‘after-the-fact’ method using the Bray–Curtis distance, and, for the Bray–Curtis ordination, the proportion of variance represented was the result of a built-in, considered a logical way of evaluating it with residual distances over the original ones (McCune and Grace 2002). We used the non-parametric Kruskall–Wallis analyses to test for overall differences among livestock systems, in terms of the scores of each sampled site along the ordination axes. Pair-wise non-parametric comparisons identified homogeneous groups of livestock systems along the axes.

Finally, to identify which species of each biological group best characterised each livestock system (third research objective) we performed an indicator species analysis (Dufrene and Legendre 1997). Indicator species analysis generates indicator values (IV) on the basis of SF and relative abundance in each type of livestock system, according to the equation:

$$IV_{ij} = A_{ij} \times B_{ij} \times 100$$

$A_{ij} = SF_{ij}/SF_i$ and $B_{ij} = NT_{ij}/NT_j$, where A_{ij} is the frequency of i species in j livestock system (SF_{ij}), in the frequency of i species on all sites (SF_i); and B_{ij} is the number of sites of j livestock system where i species is present (NT_{ij}), over the total number of sites of j livestock system (NT_j). IV range from 0% to 100%, reaching 100% when i species have their maximum frequency (i.e. when SF_i is 9) at every sites of j livestock system, and it is present at no site of another system. We performed a Monte Carlo test (4999 permutations) to assess statistical significance of observed values.

Infostat (Di Rienzo *et al.* 2011) was used to perform ANOVA, Kruskal–Wallis and Wilcoxon analyses, Tukey and pair-wise comparisons and to check normality and homogeneity of variance. We used SAM version 2.0 (Rangel *et al.* 2006) to perform correlograms. Indicator species analysis and ordination analyses were executed with PC-ORD 5.0 (McCune and Mefford 1999).

Results

The three livestock systems included in this study did not differ significantly in measures of local diversity. Comparison of Inverse Simpson index among systems showed no differences in the diversity of birds and large- and medium-sized mammals

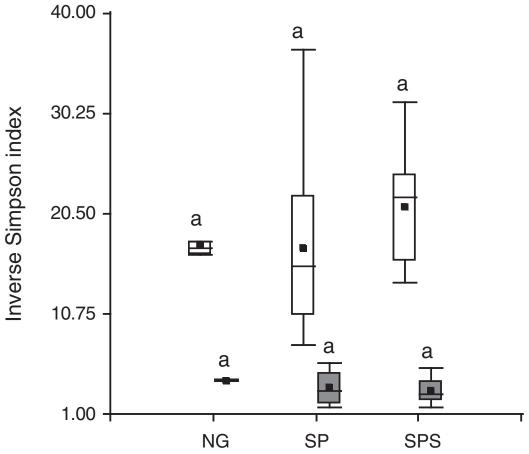

Fig. 2. Inverse Simpson diversity indexes of birds (empty boxes) and large mammals (grey boxes) in three livestock systems of open environments (NG = natural grasslands; SP: sown pastures; SPS: silvopastoral systems). Boxes show median and mean value (line and point inside boxes, respectively), quartiles and extreme values. Boxes with the same letter do not statistically differ ($P < 0.05$) in terms of their mean values.

(C_{inv}, $P = 0.29$ and $P = 0.37$, respectively; Fig. 2). The ranges of the index values varied among land-use categories, with sown pastures having the largest variation and natural grasslands the smallest (Fig. 2).

Sites, segregated along three axes in the ordination diagram based on the frequency of bird species, represented 83% of the variance (Fig. 3*a*, *b*). The first axis grouped sown pastures with silvopastoral systems (low scores), clearly away from natural grasslands (high scores). This segregation was evident in pair-wise comparisons resulting from the Kruskal–Wallis analysis ($H = 10.28$, $P = 0.006$). The second axis did not significantly separate sites from different types of livestock system ($H = 3.19$, $P = 0.20$). The third axis separated sown pastures (high scores) from silvopastoral systems (low scores), with natural grasslands as part of both groups with intermediate values ($H = 14.45$, $P < 0.001$; Fig. 3*a*, *b*). Bray–Curtis ordination based on frequency of mammals segregated sites into three dimensions, representing 62% of the variance (Fig. 3*c*, *d*). The Kruskal–Wallis analysis and pair-wise comparisons showed that no axis significantly segregated livestock systems in the ordination space ($P > 0.05$). However, graphs show some segregation of natural grasslands in relation to the other systems on axes 1 and 3 (Fig. 3*c*, *d*).

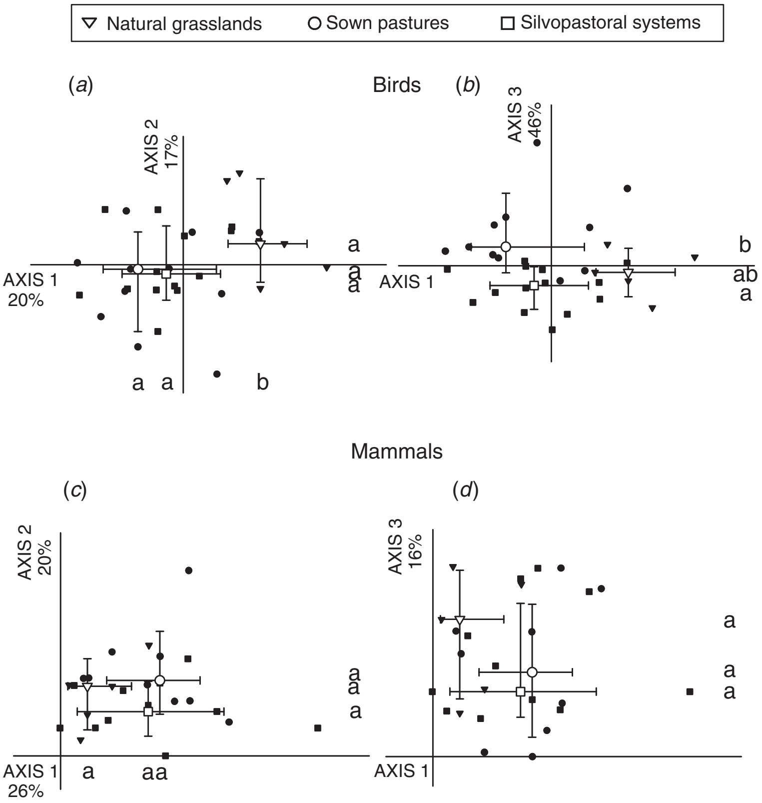

Fig. 3. (*a*, *b*) Non-metric multidimensional scaling ordination of sampled sites, based on species frequency of birds; and (*c*, *d*) Bray–Curtis ordination of sampled sites based on large mammals species (or group of species) frequency. Diagrams represent axis 1 versus 2 (*a*, *c*) and 1 versus 3 (*b*, *d*). Percent of explained variance is shown for each ordination axis. Solid symbols represent sites and empty symbols, the median values of the scores. Bars represent standard deviation from the median value. Small letters along each axis show the homogenous groups resulting from Kruskal–Wallis analyses and pair-wise comparisons (same letters indicate similar median values, $P < 0.05$).

The indicator species analysis associated nine bird species with natural grasslands, three with sown pastures and four with silvopastoral systems (Table 1). Finally, the indicator species analysis performed with mammals highlighted only *Mazama gouazoubira* and the 'Rabbits' group as having significant IV, both of them associated with natural grasslands (Table 1). The giant anteater, *Myrmecophaga tridactyla*, did not reach statistically significant values ($P = 0.19$), but had an IV four times higher in natural grasslands than in sown pastures; and was not recorded in silvopastoral systems (Appendix 1).

Discussion

Differences in habitat quality among livestock systems of open environments, at this scale of analysis and based on components of the landscape, such as native grasses species and trees, did not translate into large quantitative differences in the local diversity of birds and mammals. The large variation observed in diversity values of sown pastures and silvopastoral systems, however, suggests that probably other factors not addressed in this work (for example, distance to forests, trees density and canopy cover) were important drivers of local diversity in open areas. In contrast, natural grasslands showed an extremely low variation among sites, suggesting that biological communities in this system could be nearer to equilibrium with their environmental conditions than sown pastures or silvopastoral systems.

Livestock systems did differ markedly in the composition of bird communities. In the most important dimension, non-metric multi-dimensional scaling ordination discriminated natural grasslands from the two other systems, thus supporting our third prediction. This pattern is consistent with the observations of Macchi *et al*. (2013).

Table 1. Indicator values of bird and mammal species (or group of species) occurring under different livestock systems

Bold numbers indicate maximum IV for each species. We only report species with significative indicator value (Monte Carlo test); remaining species are reported in Appendix 1. $^*P < 0.10$; $^{**}P < 0.05$

Species	Livestock systems of open environments Natural grasslands	 Sown pastures	 Silvopastoral systems
	Birds		
Aratinga acuticaudata	**42***	22	5
Chunga burmeisteri	0	0	**36****
Coragyps atratus	13	**45****	3
Coryphospingus cucullatus	6	12	**53****
Drymornis bridgesii	**44****	3	10
Embernagra platensis	2	**48****	7
Furnarius cristatus	**56****	0	1
Hemitriccus margaritaceiventer	**50****	9	8
Patagioenas maculosa	**50****	11	33
Pyrocephalus rubinus	**31***	1	1
Polioptila dumicola	**56****	2	8
Sicalis flaveola	0	3	**47****
Stigmatura budytoides	**62****	6	9
Suiriri suiri	11	3	**51****
Turdus amaurochalinus	**36***	3	7
Zenaida auriculata	1	**47****	25
	Mammals		
Rabbits	**45****	1	3
Mazama gouazoubira	**46***	12	28

Natural grasslands had more indicator species than the other livestock systems; *Furnarius cristatus*, for example, had almost exclusive records on natural grasslands (except for one record in silvopastoral systems). As an indicator species is the result of a higher frequency and relative abundance in one category with respect to the others, this result highlights natural grasslands as an environment with original contributions to regional biodiversity (Macchi *et al*. 2013; Grau *et al*. 2015). However, every species, acting as an indicator of natural grasslands, also inhabits semi-arid Chaco forests (Marinaro *et al*. 2014). This convergence in the composition of the bird community suggests some similarity in resources supplied by natural grasslands and forests, which has been observed in other tropical ecosystems (Cárdenas *et al*. 2003; Giraldo *et al*. 2011). In contrast, indicator species of sown pastures are very representative of open environments and, therefore, they are not usually found in forests. Finally, bird indicator species for silvopastoral systems are typical of semi-open areas, which would explain their presence in these systems with components of both pasture and forest covers Additionally *Chunga burmeisteri*, an indicator species of silvopastoral systems, was not recorded in natural grasslands nor sown pastures; probably because this species nests exclusively on trees, despite being a terrestrial predator (Boyle 1917).

The mammal community appeared to be less sensitive than the bird community to differences in the livestock systems of open environments in the semi-arid Chaco, as ordination did not produce significant results for mammals. However, in axes 1 and 3 some segregation of natural grasslands in respect of the other livestock systems was observed. More clearly, the indicator species analysis associated the grey brocket *Mazama gouazoubira*, 'Rabbits' (highly consumed as bushmeat) and (although non-significantly) *Myrmecophaga tridactyla* to natural grasslands; which had very low IV in sown pastures and was not recorded in silvopastoral systems. These results suggest that natural grasslands provide much better resources and habitat quality for these species, in addition to allowing their movement, camouflage and reproduction. *Myrmecophaga tridactyla* is a high conservation value species because of their low populations in the dry Chaco ('vulnerable' category in the IUCN red list; Miranda *et al*. 2014), and it clearly prefers natural grasslands rather than closed forests (Bellis *et al*. 2008; Cuéllar Soto 2011; Torres *et al*. 2014). According to the pattern observed here, we argue that for large- and medium-sized mammal, the presence of native grasses is more important than the presence of trees (prediction 1). Although not reaching statistical significance, other mammal species also differed in their IV among livestock systems, with unique records in some systems and absences in others. For example the yellow armadillo (*Euphractus sexcinctus*) was never recorded in natural grasslands. The lack of statistical significance in mammals could be also be attributed to the arrangement in groups of several species due to limitations in the methodology; possibly with diverging responses to the factors influencing livestock systems.

Overall, natural grasslands showed the highest number of associated species, and seem to be also the most distinctive system in terms of their biological communities. As livestock systems are particularly associated with different land-tenure regimes, understanding the interactions between production systems and biological communities is key for implementing management policies that minimise land-use effects on biodiversity (Mattison and Norris 2005; Fischer *et al*. 2008). Additionally, the influence of other variables, such as tree density and distance to forests, could be explored to clarify the influence of livestock systems and even reveal new patterns. Current initiatives of land-use and biodiversity conservation in the Chaco not only neglect natural grasslands as a conservation target but may also redirect anthropic pressure for productive purposes on these environments by more intensively excluding agriculture expansion into forested landscapes (Miles and Kapos 2008). Locally, protected areas do not seem to have management policies to ensure their conservation (Cardozo *et al*. 2011; Grau *et al*. 2015). Here, we contribute to the knowledge of their diversity of birds and mammals, and highlight their importance in terms of some particularly threatened species. As dry Chaco are historically conformed as a mosaic of patches of forests and grasslands, many species probably not only use but depend on grasslands (Torres *et al*. 2014; Grau *et al*. 2015). Based on our results, we support the need to include these environments as an explicit conservation target in land-use and conservation policies (Grau *et al*. 2015).

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Appendix 1. Species (or group of species) of birds and large mammals with non-significant IV ($P > 0.10$)

Species	Livestock systems of open environments		
	Natural grasslands	Sown pastures	Silvopastoral systems
	Birds		
Agelaiodes badius	2	34	22
Amazona aestiva	14	7	23
Ammodramus humeralis	0	2	18
Buteo magnirostris	4	15	8
Cacicus solitarius	13	23	4
Campephilus leucopogon	10	1	5
Caracara plancus	0	33	28
Columbina picui	6	23	41
Coryphistera alaudina	9	8	32
Cranioleuca pyrrhophia	6	1	16
Crypturellus tataupa	0	3	14
Cyanocompsa brissoni	5	15	3
Falco femoralis	8	1	17
Falco sparverius	0	30	19
Furnarius rufus	15	12	8
Guira guira	11	27	5
Icterus cayanensis	22	5	4
Lepdicolaptes angustirostris	29	7	28
Leptotila verreauxi	12	6	17
Machetornis rixosus	0	22	11
Melanerpes cactorum	6	3	21
Melanerpes candidus	0	4	8
Milvago chimachima	0	6	19
Mimus triurus	14	14	20
Molothrus bonariensis	0	21	18
Myiopsitta monachus	8	34	36
Nothoprocta cinerascens	0	5	26
Ortalis canicollis	6	6	9
Paroaria coronata	3	11	35
Parula pitiayumi	13	13	12
Patagioenas cayenensis	9	8	1
Patagioenas picazuro	18	7	35
Pitangus sulfuratus	4	17	31
Poospiza melanoleuca	39	4	17
Pseudoseisura lophotes	21	1	11
Saltator aurantiirostris	33	0	19
Saltatricula multicolor	23	8	23
Serpophaga subcristata	18	7	25
Sicalis luteola	0	0	29
Tamnophilus caerulescens	23	6	4
Thraupis bonariensis	22	5	10
Troglodytes aedon	26	13	43
Xolmis coronata	7	12	3
Xolmis irupero	6	8	39
Zonotrichia capensis	10	12	34
	Mammals		
Canidae	22	32	24
Euphractus sexcinctus	0	16	4
Myrmecophaga tridactyla	27	7	0
Small Dasypodidae	10	24	22
Small Felidae	10	38	8
Procyon cancrivorus	10	3	2
Puma concolor	10	0	9
Tayassuidae	36	18	9