

The impact of feral horses on grassland bird communities in Argentina

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Abstract

The impact of introduced herbivores on the composition and structure of plant communities has been widely studied. However, little is known about how they affect wildlife. We studied the impact of feral horses under different grazing regimes on the communities of birds in a nature reserve in the Pampas grasslands in Argentina. The areas that had predominantly tall grass (enclosures and areas of moderate grazing intensity) showed the greatest species richness and total abundance of birds. Some species, e.g. the southern lapwing (*Vanellus chilensis*), were associated with the presence of horses, while others, e.g. pipits (*Anthus* spp), were more common in lesser grazed areas. The presence of feral horses was associated with an increase in the rate of predation of eggs which varied from 12.5% within the enclosures to 70% in grazed areas. It is suggested that the increase in predation rate was due to the increased visibility of the nests and an increase in the density of opportunist carnivores. Small areas of grassland in a good state of conservation could serve as sources that would maintain communities of birds in the more transformed sections.

INTRODUCTION

It is internationally recognised that the introduction of species by man, whether directly or indirectly, to regions far from their centres of origin, is one of the main causes of biodiversity loss (Coblentz, 1990; Macdonald, 1990; Waage & Berks, 1997; Baskin, 2002). There are practically no natural areas left in the world that are free from this problem (Macdonald & Frame, 1988; Usher, 1988). Invasive species alter biogeochemical cycles and may act as competitors, predators, parasites or pathogens of native species affecting their survival (Diamond & Case, 1986; Vitousek, 1990; Usher, 1991; Cronk & Fuller, 1995).

In particular, it is known that introduced herbivores can cause significant impacts on biological diversity (Lever, 1994). Grazing produces changes in the composition of plant communities, modifies the horizontal structure of the vegetation creating open shrublands and increasing the percentage of bare ground, which in turn causes changes in the associated wildlife (Holechek, Pieper & Herbel, 1989; Hobbs, 2001). This has been shown for grassland birds, where the presence of cattle implies the replacement of entire sets of species with different tolerances to grazing (Cody, 1985; Baines, 1988; Bock & Bock, 1988; Sutter, Troupe & Forbes, 1995; Chamberlain

et al., 2000; Gates & Donald, 2000; Söderström, Pärt & Linnarsson, 2001) and even the disappearance of the most sensitive species, usually replaced by generalist birds with wider ranges of distribution (Knopf, 1996). Among the mechanisms responsible for this impact, it is generally accepted that herbivores can affect the breeding success of birds, both directly by reducing the number of available nesting sites or by nest trampling (Barker *et al.*, 1990; Grant *et al.*, 1999) and indirectly by exposing them to predators (Martin, 1993; Brua, 1999; Willson *et al.*, 2001).

Invasive herbivores differ from the typical situation of cattle since they are free-ranging and non-managed, but despite this they have received very little attention. Only scant information is available concerning the impact of feral horses on birds (Dobbie, Berman & Braysher, 1993: 33), although Levin *et al.* (2002), working in salt marshes in North Carolina, reported the replacement of dominant species and a subsequent increase in the diversity of foraging birds in sites subject to grazing by feral horses.

In Argentina, studies of the effect of exotic wild herbivores on natural ecosystems focus on their impact on the vegetation (Veblen *et al.*, 1992; Raffaele & Veblen, 2001; Chaneton *et al.*, 2002). Recently Vázquez (2002) discussed multiple effects associated with the presence of alien mammalian herbivores in the temperate forests of Argentina and Chile, pointing out their impact as competitors, as prey for opportunist predators and as carriers of transmittable diseases to the native fauna.

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In this paper we have compared the communities of birds found in sites of high grazing intensity by feral horses with closed-off areas or sites of moderate grazing intensity in a protected area of natural grassland in the Pampas. We also looked at the possible role of nest predation as the selective force responsible for the observed variations.

STUDY AREA

Our field work was carried out in the Ernesto Tornquist Provincial Park (ETPP), a nature reserve in the south of Buenos Aires province, between 38°00'–38°07' S and 61°52'–62°03' W. The reserve was established in 1940 and covers 67 km² of valleys and hills ranging from 450–1000 m in height. It has a temperate climate (Burgos, 1968) with a mean annual temperature of 14°C and mean annual rainfall of 800 mm (936 mm during the study year). The vegetation type is grass steppe dominated by *Stipa* and *Piptochaetium* (Cabrera, 1976; Frangi & Bottino, 1995). The reserve is one of the last relicts of more or less well conserved areas in the Pampas ecoregion where several endemic taxa can be found (Long & Grassini, 1997).

In 1942, five horses were introduced into the reserve, the population grew at a fast rate and has reached approximately 30 horses/km² at the present time (Scorrolli, 1999). Their impact on the vegetation is notable, by reducing the biomass and replacing species (Barrera, 1991; Kristensen & Frangi, 1992), but their effects on wildlife have not yet been evaluated.

MATERIALS AND METHODS

Bird communities

We carried out surveys in eight sectors of valleys under different grazing regimes: one fenced area (enclosure) that had not been grazed for 6 years (area of exclusion), three areas with moderate grazing intensity and four open sectors where feral horses are free to graze. The minimal distance between plots was 300 m. The characteristics of each study plot are shown in Table 1.

In each of these areas we recorded the birds present using strip transects (Eberhardt, 1978) that were 400 m long and 50 m wide for a total of 44 transects in the enclosure and moderately grazed areas and 32 in the intensively grazed areas (36 transects were studied during the winter of 2001 and 40 in the summer of 2001). The transects were placed randomly at intervals of more than 50 m and more than 50 m from the streams to avoid the effect of the presence of birds typical of that habitat. We used 7 × 50 binoculars for all bird observations (Bibby, Burgess & Hill, 1993; Ralph *et al.*, 1994).

The vegetation in each sample was recorded using 4 m² quadrats, the first being placed 50 m from the start of the transect and then at 100 m intervals. We estimated the percentage of vegetation cover in three height categories (up to 15 cm, between 15–30 cm and more than 30 cm), the percentage cover of woody species and tussock

paspalum (*Paspalum quadrifarium*) and the percentage of bare soil. We used average values from this data to characterise the vegetation of each transect. The ground cover data for tussock paspalum were noted separately, since this species creates a special habitat with its characteristic group of birds (Comparatore *et al.*, 1996).

In order to ordinate the samples by means of their vegetation structure we carried out a principal components analysis (PCA) using the matrix of variance–covariance of the variables of the vegetation measured in each transect, having previously modified the data by the arcsine of the square root (Gauch, 1982).

Using the percentage cover data for the three height categories and the percentage of bare soil we calculated the habitat diversity for each transect using the Shannon–Weiner index (H: Moore & Chapman, 1986). We compared the diversity of habitats in areas with high grazing intensity and those with moderate grazing or no grazing using a *t*-test.

We calculated the total density of birds, species richness and diversity for each transect and studied their correlations with the six variables of vegetation structure and with habitat diversity. We used analysis of variance to determine the differences between bird density, richness and diversity in the eight sectors under study, in the two different seasons and in order to compare the corresponding averages in areas under intensive grazing with those of moderate or no grazing. We used Bartlett's test to verify the homogeneity of variance, using the natural log values when necessary.

We compared the density of the most common birds in all three grazing regimes using a *t*-test. We carried out a correlation analysis to evaluate the relationship between the first principle component of vegetation structure and the density of these bird species. We carried out a PCA using the matrix of variance–covariance of the densities of bird species in each transect, leaving out the species that occurred in less than 5% of the samples.

Egg predation

We selected two replicates from each grazing regime to compare egg predation in areas of intensive grazing by feral horses with that in sectors of moderate or no grazing. We placed 10 quail (*Coturnix coturnix japonicus*) eggs, which are similar in size to the eggs of typical Pampas grassland birds (De la Peña, 1987), in each of two 100 m transects in each replicate. We put the eggs directly on the ground underneath the bush or clump of grass closest to the sampling point, in order to mimic the nesting habitat described by De la Peña (1987) and to avoid unconscious bias in their placement. We visited the transects every 3 days over a period of 14 days and recorded the number of broken or missing eggs per transect and we compared the percentage of predation in each habitat using a *t*-test.

The use of quail eggs in testing the risk of egg predation in wild birds is controversial. Some authors consider it a reliable indicator (Gottfried & Thompson, 1978; Loiselle & Hoppes, 1983; Martin, 1987), while others suggest that it can lead to erroneous conclusions

Table 1. Characteristics and grazing history of each study plot

Study plot	0–15 cm vegetation strata	15–30 cm vegetation strata	+30 cm vegetation strata	Woody species	Tussock paspalum	Bare ground	Diversity of habitats	Area (km ²)	Grazing history and current density of horses
E	4.58 (±3.55)	8.72 (±6.32)	86.25 (±9.32)	3.85 (±4.66)	2.08 (±7.22)	0.73 (±1.35)	0.67 (±0.34)	0.2	Under horse grazing since 1942, enclosed for the last 6 years.
M 1	29.38 (±11.69)	22.03 (±8.90)	45.55 (±16.32)	7.27 (±5.19)	1.41 (±3.26)	3.05 (±3.95)	1.54 (±0.21)	0.70	Under horse grazing since 1942. Currently holds a herd of 12 adult horses (17 horses/km ²).
M 2	14.91 (±9.82)	17.86 (±9.14)	64.55 (±19.12)	3.48 (±3.93)	5.54 (±8.70)	2.05 (±5.23)	1.23 (±0.38)	6.00	Currently holding a herd of 10 adult horses and 60 cattle (1.6 and 10 herbivores/km ² , respectively).
M 3	60.88 (±18.48)	18.63 (±9.19)	19.13 (±13.15)	3.75 (±3.33)	5.88 (±8.50)	1.13 (±3.56)	1.23 (±0.36)	0.50	Under horse grazing since 1942, enclosed but partially accessible to horses because of failures in fence maintenance. Current approximate density: 6 horses/km ² (pers. obs.).
I 1	84.86 (±9.71)	7.08 (±7.86)	4.03 (±4.18)	3.33 (±4.33)	0.14 (±0.42)	3.89 (±6.69)	0.70 (±0.34)	0.15	Under horse grazing since 1942, currently holding 30 horses/km ² , with a 1:1 sex ratio, and a population structure of 74% adults, 11% yearlings and 15% foals (A. Scorolli, pers. comm.).
I 2	86.82 (±3.85)	3.75 (±1.94)	5.57 (±5.49)	6.02 (±5.09)	0.00 (±0.00)	3.86 (±4.13)	0.69 (±0.16)	0.15	
I 3	87.40 (±7.45)	6.04 (±5.27)	3.54 (±2.71)	3.23 (±2.03)	0.00 (±0.00)	3.23 (±4.04)	0.67 (±0.28)	0.15	
I 4	86.72 (±7.82)	7.50 (±7.62)	3.91 (±2.05)	3.13 (±1.49)	0.94 (±2.65)	1.88 (±2.91)	0.67 (±0.23)	0.15	

E, enclosure – no grazing; M1–M3, moderate grazing; I1–I4, intensive grazing. Vegetation variables are given as % cover, except for diversity of habitats (see the text). Values in parentheses are standard deviations.

Table 2. Bird species observed during the study period

Frequency	Scientific name	Common name
Common species, seen in more than 5% of the transects	<i>Nothura maculosa</i>	Spotted tinamou
	<i>Milvago chimango</i>	Chimango caracara
	<i>Vanellus chilensis</i>	Southern lapwing
	<i>Speotyto cunicularia</i>	Burrowing owl
	<i>Hymenops perspicillatus</i>	Spectacled tyrant
	<i>Anthus spp</i>	Pipit
	<i>Sicalis luteola</i>	Grassland yellow-finch
	<i>Zonotrichia capensis</i>	Rufous-collared sparrow
	<i>Embernagra platensis</i>	Great pampa-finch
	<i>Pseudoleistes virescens</i>	Brown-and-yellow marshbird
	<i>Sturnella loyca</i>	Long-tailed meadowlark
Occasional species present in less than 5% of the transects	<i>Rhynchotus rufescens</i>	Red-winged tinamou
	<i>Falco sparverius</i>	American kestrel
	<i>Columba picazuro</i>	Picazuro pigeon
	<i>Zenaida auriculata</i>	Eared dove
	<i>Cyanoliseus patagonus</i>	Burrowing parrot
	<i>Colaptes campestris</i>	Field flicker
	<i>Geosita cunicularia</i>	Common miner
	<i>Muscisaxicola macloviana</i>	Dark-faced ground-tyrant
	<i>Tyrannus savana</i>	Fork-tailed flycatcher
	<i>Cistothorus platensis</i>	Grass wren
	<i>Mimus triurus</i>	White-banded mockingbird
	<i>Mimus saturninus</i>	Chalk-browed mockingbird
	<i>Carduelis magellanica</i>	Hooded siskin
	<i>Molothrus bonariensis</i>	Shiny cowbird
Species seen outside the transects	<i>Chloephaga picta</i>	Upland goose
	<i>Geranoetus melanoleucus</i>	Black-chested buzzard-eagle
	<i>Polyborus plancus</i>	Crested caracara
	<i>Spizapteryx circumcinctus</i>	Spot-winged falconet
	<i>Columba maculosa</i>	Spot-winged pigeon
	<i>Colaptes melanolaemus</i>	Golden-breasted woodpecker
	<i>Furnarius rufus</i>	Rufous hornero
	<i>Pitangus sulphuratus</i>	Great kiskadee
	<i>Pyrocephalus rubinus</i>	Vermilion flycatcher
	<i>Tachycineta leucorrhoa</i>	White-rumped swallow
	<i>Troglodytes aedon</i>	House wren
	<i>Turdus falklandii</i>	Austral thrush
	<i>Passer domesticus</i>	House sparrow
<i>Sicalis flaveola</i>	Saffron finch	
<i>Sicalis lebruni</i>	Patagonian yellow-finch	

(Willebrand & Marcström, 1988; Yahner, Morrell & Rachael, 1989; Haskell, 1995; Ortega *et al.*, 1998). In general the objections refer to the fact that different groups of predators might consume quail eggs and wild bird eggs, which would affect the relative rate of true nest predation. Differences may also arise due to the presence of adult birds defending their nests and due to the fact that, in general, quail eggs are larger than the wild birds' eggs and so they are not attacked by certain groups of predators with small beaks or jaws (Pärt & Wretenberg, 2002). According to data compiled by De la Peña (1987) eight out of the 21 species of birds that nest on the ground in the grasslands in the south of the Pampas have eggs with diameters that are larger than quail eggs, while six of the remaining 13 species have eggs just 20% smaller, thus reducing the described problem.

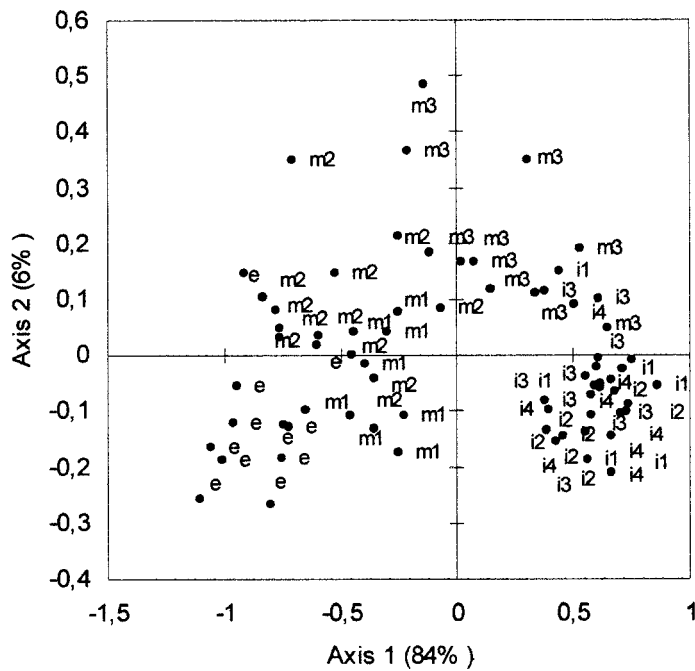
RESULTS

Bird communities

We recorded a total of 40 species of birds, 25 of them within the transects and 15 in additional observations (Table 2).

The PCA carried out using the vegetation data clearly separates the enclosure from areas with a high density of horses on the first component (84% of total variance). The transects in moderately grazed areas are found in an intermediate position (Fig. 1). This separation corresponds essentially to a dominance of the taller vegetation strata in the enclosure and a greater representation of the shorter strata in the grazed areas.

The diversity of habitats was significantly greater ($t = 6.52$; $P < 0.01$) in the moderately grazed and enclosed



	PC1	PC2
0-15 cm	0.9939	-0.0033
15-30 cm	-0.3743	0.6187
+ 30 cm	-0.9910	-0.0705
Woody spp.	0.0540	-0.3431
Tussock paspalum	-0.2980	0.7169
Bare ground	0.2802	-0.3792
Eigenvalue	0.3592	0.0239

Fig. 1. Principal components analysis (PCA) ordination of transects when considering their vegetation characteristics and the correlations between the initial variables and the principal factors. e, enclosure; m1–m3, moderate grazing; i1–i4, intensive grazing.

areas ($H = 1.20$, standard error (SE) = 0.45, $n = 52$) than in the sectors of intensive grazing ($H = 0.68$, SE = 0.25, $n = 40$).

The mean density and species richness of birds showed a positive correlation with the vegetation cover in the 15–30 cm and >30 cm strata and a negative correlation with the <15 cm strata. The diversity of birds showed a negative correlation with the cover of woody species. The three variables of structure of bird communities showed a positive correlation with the cover of tussock paspalum and the diversity and density of birds showed negative correlations with the percentage of bare ground. No significant correlations were found between these parameters and the diversity of habitats (Table 3).

Considering each area separately we noted that moderately grazed areas showed slightly higher values of density, species richness and diversity of species in comparison with areas of exclusion (the enclosure) and

of high grazing intensity, although these differences were not statistically significant (Table 4).

The mean values for species richness and diversity of birds under each grazing regime did not show any significant seasonal differences ($P > 0.1$). However, the analysis of density showed a highly significant interaction ($P < 0.01$) between sectors and seasons and so an analysis of variance was repeated, separating the data for winter and spring. The moderately grazed areas and the enclosure showed higher values for richness of species and diversity during the whole year and greater density during the spring (Table 5).

We found that certain species were exclusively or significantly associated with the enclosure or with moderately grazed areas, e.g. *Sicalis luteola*, *Sturnela loyca*, *Anthus* spp and *Embernagra platensis*. However, *Vanellus chilensis* preferred habitats with a high grazing intensity, while *Pseudoleistes virescens*, *Nothura maculosa*,

Table 3. Correlation indexes between the vegetation variables and the diversity, density and species richness of birds for the total number of recordings ($n = 76$)

Vegetation variables (expressed as % cover except for diversity of habitats)	Species diversity (Shannon–Weiner index)	Density (birds/ha)	Species richness (number of species)
0–15 cm	0.06	–0.77**	–0.46**
15–30 cm	0.01	0.87**	0.46**
+30 cm	–0.05	0.69**	0.41**
Woody species	–0.38**	0.14	0.03
Tussock paspalum	0.60**	0.68**	0.64**
Bare ground	–0.37**	–0.49**	–0.21
Diversity of habitats	0.21	0.14	0.14

** , $P < 0.01$, highly significant correlation.

Table 4. Means and standard deviations for the three variables of bird community structure in the eight sectors sampled

	Density (birds/ha)		Species richness (number of species)		Species diversity (Shannon–Weiner index)	
	Mean	Standard deviation	Mean	Standard deviation	Mean	Standard deviation
E	3.32	2.08	2.09	1.04	0.82	0.70
M1	4.63	10.47	1.63	1.06	0.43	0.48
M2	2.42	1.94	2.08	1.08	1.09	0.83
M3	3.65	2.80	2.90	1.37	1.20	0.78
I1	2.00	2.76	1.67	1.03	0.50	0.82
I2	0.86	0.56	1.29	0.49	0.26	0.45
I3	1.00	0.79	1.33	0.50	0.55	0.57
I4	2.63	1.73	2.00	1.07	0.70	0.61

E, enclosure – no grazing; M1–M3, moderate grazing; I1–I4, intensive grazing.

Table 5. Results of the analysis of variance between the three parameters of bird community structure and the two grazing regimes

	Species richness (number of species)	Species diversity (Shannon–Weiner index)	Density in winter (birds/ha)	Density in spring (birds/ha)
Moderately grazed areas and enclosure ($n = 44$)	2.38 ± 0.20	0.87 ± 0.11	2.68 ± 0.59	4.52 ± 1.34
Intensively grazed areas ($n = 32$)	1.65 ± 0.16	0.48 ± 0.10	1.78 ± 0.85	1.52 ± 0.32
F	7.19	6.04	0.38	12.59
P	0.009	0.016	0.54	0.001

The mean species richness, diversity and density of birds are shown for both areas, together with their standard errors.

Hymenops perspicillatus, *Milvago chimango* and *Speotyto cunicularia* were found to be equally abundant in both habitats (Fig. 2). These results are consistent with the

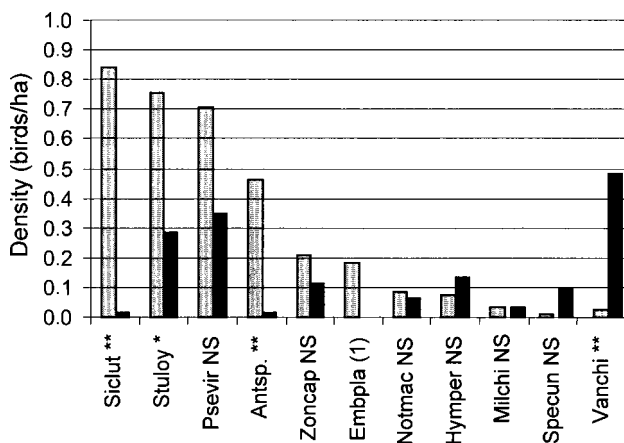


Fig. 2. Density (birds/ha) of the species recorded in more than 5% of the transects in the areas of exclusion (enclosure), moderate grazing and intensive grazing. Siclut, *Sicalis luteola*; Stuloy, *Sturnella loyca*; Psevir, *Pseudoleistes virescens*; Antsp, *Anthus* spp; Zoncap, *Zonotrichia capensis*; Embpla, *Embernagra platensis*; Notmac, *Nothura maculosa*; Hymper, *Hymenops perspicillatus*; Milchi, *Milvago chimango*; Specun, *Speotyto cunicularia*; Vanchi, *Vanellus chilensis*; *, $P < 0.05$; **, $P < 0.01$; NS, no significant difference; □, moderate grazing and enclosure; ■, intensive grazing; (1), species exclusively from moderate grazing and enclosure.

analysis of correlation between the density of each species and the first principal component of vegetation structure. Density of *Anthus* spp. was significantly associated with the values of this component corresponding to areas of exclusion (i.e. the enclosure) and moderate grazing whereas those of *Speotyto cunicularia* and *Vanellus chilensis* were associated with areas of intensive grazing (Table 6).

The PCA using the bird community data concentrates the intensively grazed samples (75% of the total) in the third quadrant of the ordination space defined by the first component (46% of the variance) and the second component (23% of the variance: Fig. 3). This separation is essentially based on the fact that these transects have lower densities of *Sturnella loyca* and *Pseudoleistes virescens*.

Egg predation

Egg predation showed a highly significant difference between both habitats ($t = 4.86$; $P < 0.01$). The sectors with higher grazing intensity showed a mean predation rate of 70% (Standard deviation (SD) = 21.60), contrasting with 12.5% (SD = 9.57) in areas of exclusion and moderate grazing.

DISCUSSION

The area that we studied shows a situation of overgrazing, which can be seen as an exception among traditional

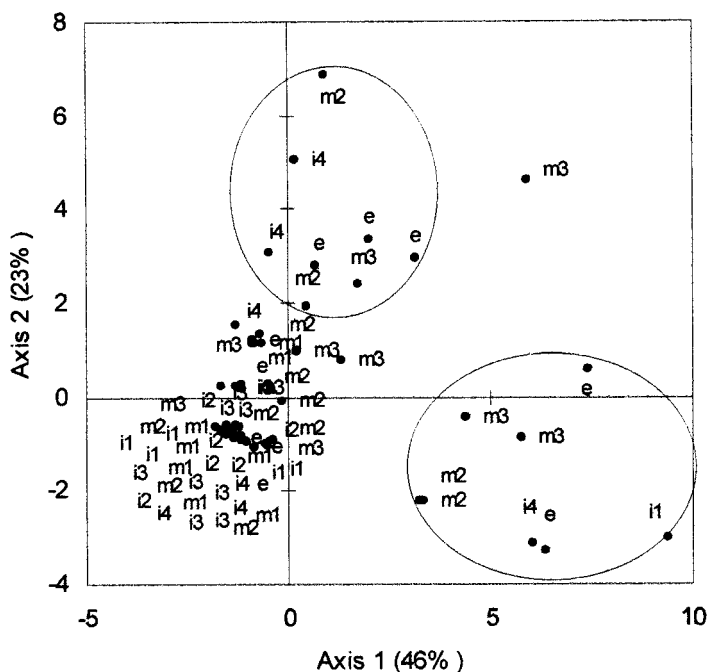
Table 6. Correlations between the density of the most common species of birds and the co-ordinates of the first axis of the PCA for vegetation data

Species	First principle component (vegetation structure)
<i>Nothura maculosa</i>	-0.08
<i>Milvago chimango</i>	-0.11
<i>Vanellus chilensis</i>	0.38**
<i>Speotyto cunicularia</i>	0.25*
<i>Hymenops perspicillatus</i>	0.02
<i>Anthus</i> spp.	-0.41**
<i>Sicalis luteola</i>	-0.11
<i>Zonotrichia capensis</i>	-0.09
<i>Embernagra platensis</i>	0.04
<i>Pseudoleistes virescens</i>	-0.16
<i>Sturnella loyca</i>	-0.21

*, significant correlation ($P < 0.05$), **, highly significant correlation ($P < 0.01$). Negative correlations represent an association with the areas of exclusion and moderate grazing while positive correlations represent an association with areas of intensive grazing. PCA, principal component analysis.

cattle raising (Knopf, 1996), but is usual in the case of invasive herbivores (Lever, 1994) where the stocking rate is usually not regulated for political or economic reasons (Dobbie *et al.*, 1993; Symanski, 1996; Beever & Brussard, 2000). In Tornquist Park this results in great changes in vegetation structure, which, in extreme cases turns into grassy meadows 15 cm high in contrast to grasslands free from exotic herbivores that are over 1 m high. These variations reflect on the richness and diversity

of birds, which are greatest in areas subject to moderate levels of grazing, slightly lower in the enclosures and much lower under high grazing pressure, coinciding with changes in the diversity of habitats. In particular it is interesting that the richness and diversity of species are slightly greater in areas of moderate grazing than in the enclosures, as predicted by the theory of intermediate disturbance (Connell, 1978). These results are interesting in the light of the discussion concerning whether or not exotic herbivores should be eradicated from protected natural areas (McCoy, 1994; Houston & Schreiner, 1995). Levin *et al.* (2002) showed many ecological effects of feral horses in salt marshes leading to conflicts between the goals of maintaining horse populations and functional marshes. Nevertheless, they also discussed the possibility that those habitats may have shared Pleistocene times with an arrangement of large herbivores that made them function more like modern horse-grazed marshes than like present-day marshes where horses are excluded. In the same fashion, Argentinean Pampas were historically subjected to grazing by native herbivores, such as deer and guanaco (Chebez, 1994) that formed remarkably large herds (Darwin, [1898] 1967; Hudson, [1918] 1963), so it is likely that typical grassland birds have evolved in habitats subjected to high to moderate grazing regimes. Could feral horses be considered equivalent to extinct or rare native herbivores? What would be an optimal density (above zero) of introduced herbivores for a grassland reserve? The answers to these questions will, no doubt, vary as a function of the objectives of each reserve, but it is important to consider at least four factors in any analysis. (1) The possible long term effects of the presence



	PC1	PC2
<i>Nothura maculosa</i>	-0.10	0.20
<i>Milvago chimango</i>	-0.05	0.00
<i>Vanellus chilensis</i>	-0.27	0.02
<i>Speotyto cunicularia</i>	-0.14	-0.03
<i>Hymenops perspicillatus</i>	-0.11	0.14
<i>Anthus</i> spp.	0.21	-0.08
<i>Zonotrichia capensis</i>	0.27	0.04
<i>Embernagra platensis</i>	0.29	0.16
<i>Pseudoleistes virescens</i>	0.97	-0.22
<i>Sturnella loyca</i>	0.41	0.91
Eigenvalues	6.20	3.07

Fig. 3. Principal components analysis (PCA) ordination of transects when considering the density of bird species present in more than 5% of the samples and the correlations between the initial variables and the principal factors. The circle on the bottom right shows the samples with greatest relative abundance of *Pseudoleistes virescens*, while the one at the top of the graph shows those with a predominance of *Sturnella loyca*. e, enclosure; m1–m3, moderately grazed areas; i1–i4, intensively grazed areas.

of alien herbivores on critical habitats e.g. waterholes in arid environments (Dobbie *et al.*, 1993), or under unusual environmental conditions e.g. severe droughts (Bock & Bock, 1998). (2) Their effects on species that are especially sensitive to grazing (Detling, 1998). (3) The dynamics of the native species on a wide geographical scale since, for example, some species only tolerate the presence of herbivores if better conserved areas exist in the same region (Wiens, 1992b: 217; Knopf, 1996). (4) The effects of exotic herbivores on the composition of the communities, rather than on species richness or diversity (Parker *et al.*, 1999).

Our work suggests that there are consistent differences in the composition of bird communities according to grazing intensity, with some species preferring areas of intense grazing and others found exclusively, or more abundantly, in areas with a low density of feral horses. It is tempting to classify these species as increasers or decreasers in response to grazing, however, this might be an over-simplification if not complimented by other information such as the history of the pasture management, or ecological factors such as soil type or rainfall (Knopf, 1996). Possible causes of differential distribution of bird species in environments with different grazing regimes include the availability of food supplies, the expected reproductive success and predation risk (Cody, 1985; Wiens, 1992a: 339).

In our case, despite the caution that must be exercised in using quail eggs as indicators of natural predation, the magnitude of increase in egg predation associated with severe grazing seems to be sufficient evidence to suggest that grazing intensity could have a significant impact on the population dynamics of grassland birds and that their survival in the area would depend on the existence of patches of vegetation free from excessive grazing.

It has been suggested that the existence of high and low quality habitats results in a sink-source metapopulation structure where the excess of individuals from productive patches emigrate into less productive ones (Pulliam, 1988). One particular point of interest would be to determine whether small, well-conserved areas of grassland act as 'sources' with a high rate of reproduction, or whether they can become 'ecological traps' for the birds. An 'ecological trap' is a site that has clues that enable it to be perceived as a high-quality site by wildlife, but which includes characteristics that increase the death rate or reduce the rate of reproduction (Ratti & Reese, 1988; Kokko & Sutherland, 2001). In the case of moderately grazed patches of grassland, the density and height of the vegetation could indicate that they are 'safe' areas for nesting, but their high proportion of border could increase the probability of nest predation. This phenomenon has been shown in numerous studies on the effect of fragmentation in forests (Wilcove, 1985; Martin, 1988; Keyser, Hill & Soehren, 1998) and more recently in natural grasslands (Phipps, 2002). In our study we observed that although the area of exclusion was as small as 20 ha, its predation rate was significantly lower than in the grazed areas. Unfortunately there were no other similar enclosed areas available that could have acted as

replicates. However, we feel cautiously optimistic about the value of small relictual areas of grassland in helping to maintain the diversity of birds in the region. The presence of feral horses could result in other indirect impacts on the breeding success of grassland birds due to the increase in the abundance of opportunistic predators and brood parasites.

The interaction between exotic herbivores, opportunistic predators and native prey has been studied by Courchamp, Langlais & Sugihara (1999) and Norbury (2001) in Australia, where high densities of rabbits are associated with an increase in the abundance of cats, foxes and ferrets, which, in turn, increases the impact of these carnivores on native species of birds and lizards. Vázquez (2002) reports many cases that indicate that the presence of exotic herbivores, such as the European hare and the rabbit, could favour native avian and mammal predators in the temperate woods of Argentina with a consequent impact on the populations of native prey. In our study area it is clear that at least one carnivorous species, the Pampas fox (*Pseudalopex gymnocercus*), takes advantage of feeding on the carrion of feral horses and, as shown by diet analysis, also preys on wild birds (Castillo, 2002). This could be the explanation for the observed increase in the abundance of foxes in the ETPP with respect to the surrounding area (Lucherini, Birocchio & Luengos Vidal, 2001). As well as the food source provided by dead horses, some native predators also feed on placentas during the breeding season and on the invertebrates that live in the manure, as in the case of the Chimango caracara (*Milvago chimango*) in Chile (Willson *et al.*, 2001).

However, the Shiny cowbird (*Molothrus bonariensis*) is a brood parasite that is relatively abundant in the area and it feeds on external parasites of feral horses (pers. obs.). The association between cattle and cowbirds has been recognised for a long time (Mayfield, 1965) and, *a priori*, it seems reasonable to assume than with any increase in the population of horses the cowbirds would be at an advantage and the breeding success of grassland birds would decrease due to parasitism of their nests. We were not able to detect a significant correlation between cowbird abundance and grazing intensity, nevertheless our experience in the area seems to indicate that the overall abundance of the species in the park has risen in the past 7–8 years, co-incidentally in with the increase in the feral horse population. Cowbirds are capable of covering long distances daily between feeding and nesting grounds, as Goguen & Mathews (2000) have shown for the brown-headed cowbird (*Molothrus ater*) and these movements could obscure differences in abundance between habitats. If our observations were correct, brood parasitism could be a long-distance edge effect that must be carefully evaluated when determining the conservation value of small areas of grassland under moderate grazing regimes.

In conclusion, the importance of analysing the problems caused by biological invasions by means of studies in areas with and without the invasive species has to be emphasised, as does that of experimental analysis of their effects on parameters at the level of the individual

organisms as well as of populations (Parker & Reichard, 1998; Parker *et al.*, 1999; Byers *et al.*, 2002). This approach would help to set research priorities concerning the impact of the invader, to adjust the necessary management interventions so as to reduce their impact and to gain objective information to support the control strategy from a political point of view.

Our results emphasise, moreover, the relevance of managing herbivore stocking rates for the conservation of grassland bird communities and, fundamentally, the importance of the preservation of patches with moderate grazing regimes which would act as a 'source' areas for reproduction, and repopulating neighbouring areas.

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