

## The influence of fire and livestock grazing on the assemblage of non-flying small mammals in grassland-Araucaria Forest ecotones, southern Brazil

Ezequiel Pedó<sup>1, 4</sup>; Thales R. O. de Freitas<sup>1, 3</sup> & Sandra M. Hartz<sup>1, 2</sup>

<sup>1</sup> Programa de Pós-Graduação em Ecologia, Universidade Federal do Rio Grande do Sul. Caixa Postal 15007, 91501-970 Porto Alegre, RS, Brazil.

<sup>2</sup> Departamento de Ecologia, Instituto de Biociências, Universidade Federal do Rio Grande do Sul. Caixa Postal 15007, 91501-970 Porto Alegre, RS, Brazil.

<sup>3</sup> Departamento de Genética, Instituto de Biociências, Universidade Federal do Rio Grande do Sul. Caixa Postal 15007, 91501-970 Porto Alegre, RS, Brazil.

<sup>4</sup> Corresponding author. E-mail address: ezequiel.pedo@gmail.com

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**ABSTRACT.** Grazing and fire, used in pasture regeneration practices, are inter-related activities in livestock management. Both activities affect habitat characteristics by changing the structure of the herbaceous and shrubby vegetation, reducing their biomass and litter cover. This study evaluated the effect of fire and livestock grazing on the assemblage of non-flying small mammals in grassland-Araucaria forest ecotones in southern Brazil. We compared four areas frequently affected by livestock management with four protected areas. Surveys were carried out in four sampling periods, one in each season of 2004. We captured a total of 325 individuals from 12 species of rodents – *Akodon montensis* Thomas, 1913, *Akodon paranaensis* Christoff, Fagundes, Sbalqueiro, Mattevi e Yonenaga-Yassuda, 2000, *Akodon* sp. 1, *Akodon* sp. 2 (2n = 34), *Brucepattersonius iheringi* (Thomas, 1896), *Delomys dorsalis* (Hensel, 1872), *Oligoryzomys flavescens* (Waterhouse, 1837), *Oligoryzomys nigripes* (Olfers, 1818), *Oxymycterus nasutus* (Waterhouse, 1837), *Scapteromys* sp. (2n = 34), *Sooretamys angouya* (Fischer, 1814) and *Thaptomys nigrita* (Lichtenstein, 1829) – and two species of marsupials – *Monodelphis dimidiata* (Wagner, 1847) and *Philander frenatus* (Olfers, 1818) –, in a total effective effort of 5254 traps.day<sup>-1</sup>. The abundance, biomass and diversity of non-flying small mammals were significantly higher in the protected areas than in those affected by fire and livestock grazing. Species strictly associated with grassland habitats were not found in the impacted area. These results indicate that the presence of herds of domestic ungulates negatively affect the assemblage of non-flying small mammals in grassland and Araucaria forest areas in southern Brazil.

**KEY WORDS.** Atlantic Forest; Campos; Didelphimorphia; habitat loss; Rodentia.

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The presence of large domestic herbivores in a landscape is generally associated with understory and litter cover reduction and soil compaction, resulting in significant environmental impact that can affect ecosystem nutrient cycle and energy flow (FRANK & EVANS 1997, FRANK & GROFFMAN 1998). In the northwestern United States, grazing has not been found to have a significant effect on bird species, but seems to negatively affect the assemblage of non-flying small mammals, which have more restricted mobility (MOSER & WITMER 2000). Even though richness, abundance and diversity of this group are lower in grazed areas when compared with non-grazed ones (MOSER & WITMER 2000), non-flying small mammal species often respond differently to grazing. Species that have more strict resource requirements (habitat specialists) tend to be more negatively affected than habitat generalists (NUPP & SWIHART 2000). Habitat loss is considered the major threat to biodiversity conservation in terrestrial ecosystems around the world (BROOKS *et al.* 2002), and species extinc-

tions in grasslands negatively affect ecosystem stability and productivity (TILMAN & DOWNING 1994, TILMAN *et al.* 1996).

The Araucaria forest (mixed ombrophilous forest, sensu TEIXEIRA & COURA NETO 1986) is physiognomically characterized by the dominance of *Araucaria angustifolia* (Bertol.) Kuntze. It is part of the Atlantic Forest domain, one of the top five biodiversity hotspots in the world, and one of the most threatened ecosystems (MYERS *et al.* 2000), with 11.4% to 16% of its original cover currently remaining (RIBEIRO *et al.* 2009). Araucaria forests constitute the main forest type on the highland plateau in southern Brazil at altitudes above 500m a.s.l.; this forest often forms mosaics with natural grasslands (DUARTE *et al.* 2006). The natural succession of grasslands by Araucaria forests, which started around 1200 years ago (BEHLING *et al.* 2004), has been hindered by human impact, particularly anthropogenic fire. In southern Brazil, grasslands are often subjected to periodic burnings during the process of pasture regeneration for livestock use (BEHLING

& PILLAR 2007). Fire affects habitat dynamics radically by removing virtually all plant biomass from the soil, and by creating a positive feedback mechanism that increases susceptibility to fire and fire intensity (COCHRANE & SCHULZE 1999, COCHRANE *et al.* 1999). This particular disturbance causes a sudden but relatively short change in the dynamics of non-flying small mammal populations (OJEDA 1989, FA & SANCHEZ-CORDERO 1993), favoring some species at the expense of others (PRICE & WASER 1984, VIEIRA & MARINHO-FILHO 1998, VIEIRA 1999, SIMON *et al.* 2002, FIGUEIREDO & FERNANDEZ 2004). Apparently, fire causes no significant direct mortality on these animals. Instead, it compels non-flying small mammals to flee (VIEIRA & MARINHO-FILHO 1998) or to find protection in subterranean dens (SIMONS 1991).

The aim of this study was to assess the influence of habitat degradation by fire and grazing on the assemblage of non-flying small mammals in grassland-Araucaria forest ecotones in southern Brazil.

## MATERIAL AND METHODS

This study was carried out in 2004, in the municipality of São Francisco de Paula, in the northwestern region in the state of Rio Grande do Sul, southern Brazil (Fig. 1). According to TEIXEIRA & COURA NETO (1986), this region comprises three phytoecological zones: grassy-woody savanna (grasslands) and mixed ombrophilous forest (Araucaria forest), located in higher sections of smoother relief, and dense ombrophilous forest (Atlantic forest), found in steep scarp slopes.

Local climate is temperate ('Cfb' type, according to KÖPPEN (1948)) characterized by well-distributed rainfalls throughout the year and average temperature lower than 22°C in the warmest summer month (KÖPPEN 1948). The average annual number of frost days is 21 to 26, and short-duration snowfall may occasionally occur in June, July and August (MORENO 1961). According to data collected between 1931 and 1960 at the São Francisco de Paula Meteorological Station (29°20'S and 50°31'W, 912 m), average annual rainfall is 2253 mm and average annual temperature is 14.5°C (average maximum temperature: 20.3°C; average minimum temperature: 9.9°C), reaching 34°C in December and -6.5°C in July (BRASIL 1969). There is a considerable surplus of water throughout the year (BRASIL 1969). Dense fog and orographic rain are also very common.

We selected eight areas for sampling, located between 29°27'58"-29°29'17"S and 50°12'0"-50°13'26"W. A control and a treatment group were defined, each with four areas. The control group has been protected from livestock grazing and grassland fire since 1994 (Centro de Pesquisas e Conservação da Natureza Pró-Mata, with an area of ca. 2500ha, hereafter named 'protected area'), while the treatment group is located in a ranch (near the protected area) subject to extensive management of cattle and pig herds, and periodic grassland burning (every one or two years) (Fazenda Três Estrelas, with an area of approximately 500ha, hereafter named 'impacted area').

One sampling area in both protected and impacted sites was located in grassland-continuous Araucaria forest ecotones, while the other three areas in each group were located in ecotones between grasslands and Araucaria forest patches (Fig. 1). The three forest patches from the protected area had 0.15, 0.22 and 0.24ha, while patches from the impacted area had 0.12, 0.22 and 0.36ha. Even though selective logging of native trees has happened in both areas in the past, logging of native trees has not taken place in the last 20 years.

A detailed description of the physiognomy of the vegetation of the protected area can be found in OLIVEIRA & PILLAR (2004) and DUARTE *et al.* (2006).

We carried out surveys of non-flying small mammals in four campaigns, one in each season of 2004. Since nights with high luminosity are associated with changes in wildlife activity patterns, which are interpreted as a response to increased predation risk, campaigns were carried out in periods when luminosity was lower, in the same lunar phase (KAUFMAN & KAUFMAN 1982, PRICE *et al.* 1984, BOWERS 1988, BROWN *et al.* 1988, BITTENCOURT *et al.* 1999). Trapping efforts per campaign totaled fourteen, 24-hour periods (except in winter, with only three periods). The first seven consecutive periods were spent in both grassland-continuous Araucaria forest ecotones, while the remaining seven were spent in the six grassland-Araucaria forest patch ecotones.

We used 144 traps (cage type, 30x14x14 cm) in each 24-hour period (hereafter day). Each trapping station was placed 15 m apart from the others and had two traps placed at ground level. Half of the stations were placed in the grassland and the other half in the Araucaria forest. Stations were allocated equally between the impacted and protected areas. Sampling in both areas was simultaneous, and trapping effort per area was the same (3240 traps.day<sup>-1</sup>). Total trapping effort (for both areas) was 6480 traps.day<sup>-1</sup>. Traps were baited with corn and a mixture of wheat flower, peanut sweet (roasted and crushed peanuts mixed with sugar and manioc flour) and fish.

Three transects were distributed in each grassland-continuous Araucaria forest ecotones. Each transect had 12 trapping stations, totaling 36 stations. In grassland-Araucaria forest patch ecotones, we established only two transects. Number of trapping stations per transect, in this case, was four, six or eight, according to patch size. In order to avoid problems in comparing different patches, we selected six Araucaria forest patches based on their similarity (shape and size), leaving treatment (grazing and fire present or absent) as the main difference between them.

Captured individuals were identified, marked by ear puncturing, measured (body and tail length), weighted and sexed. We collected at least one individual of each captured species for deposition in the mammal collections of the Museu de Ciências e Tecnologia of the Pontifícia Universidade Católica do Rio Grande do Sul (MCT-PUCRS) and of the Museu Nacional do Rio de Janeiro (MNRJ).

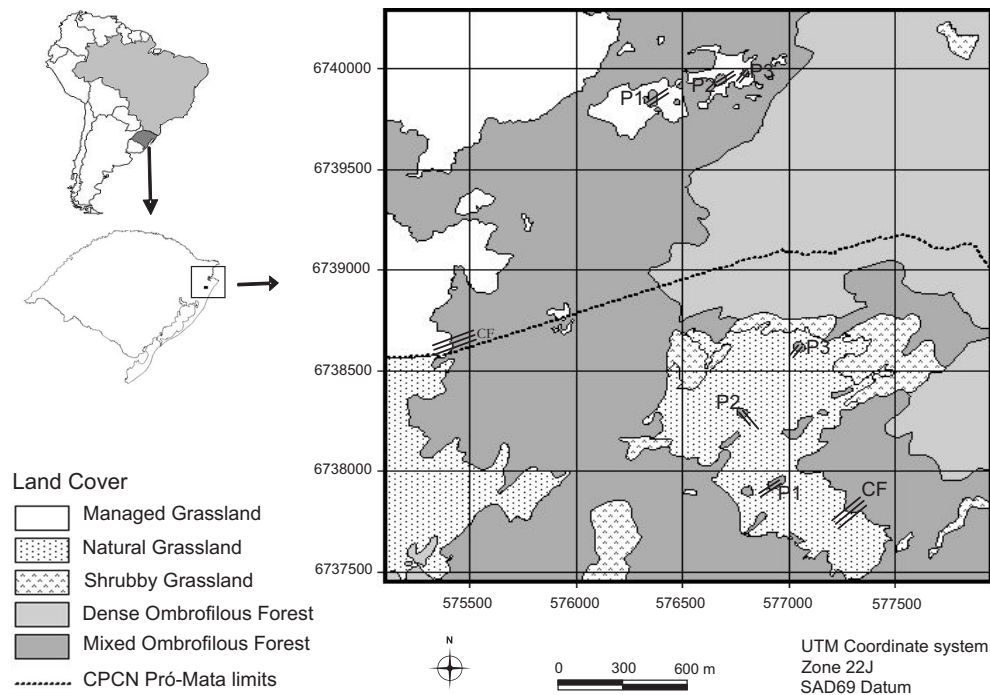


Figure 1. Study site location (land use and land cover maps), with use of ArcView GIS 3.2). Abbreviations of sampling areas: (CF) transects in grassland-continuous forest ecotones; (P1, P2 and P3) transects in different grassland-forest patch ecotones). The impacted area (Fazenda Três Estrelas) is located northwards from the divider indicated in the map while the protected area (CPCN Pró-Mata) is southwards.

We compared impacted and protected areas using abundance (number of individuals) and biomass (in grams) per mammal species. Data matrices were generated using an abundance index calculated as the number of individuals per species divided by trapping effort in the four sampling campaigns, in each sampling unit (i.e. trapping station).

To compare the composition of non-flying small mammals between areas under different management regimens (i.e. impacted or protected), we used MULTIV version 2.3.17 (PILLAR 2005), a multivariate statistics software that generates similarity statistics and undertakes analysis of variance using randomization tests (non-parametric hypothesis testing), as proposed by PILLAR & ORLÓCI (1996). In the analysis of variance, total variation is partitioned into 'between-group' and 'within-group' variation (MANLY 1997). The purpose of variance partitioning is to assess, by means of the probability (P) obtained in randomization tests, the magnitude of 'between-group' variation, and whether groups differ or not. Partitioning was performed by means of the sum of squared Euclidean distances between sampling units. The randomization test generates a probability (P) ( $Qb^o \geq Qb$ , where the random sum of squares is greater than or equal to the observed sum of squares) used to reject or accept the null hypothesis ( $H_0$ ). The null hypothesis is rejected if P is lower than or equal to  $\alpha$  (0.1), and is accepted if there is

no significant difference between protected and impacted areas. To test the hypothesis, 1000 iterations were applied.

Trapping effort per sampling unit was uneven throughout the year, because traps were occasionally disarmed by weather, cattle or predators – mainly the crab-eating fox, *Cerdocyon thous* (Linnaeus, 1766). To overcome this problem, we estimated the average number of individuals per trap.day<sup>-1</sup> in each sampling unit for comparison between the two areas, and probability (P) was estimated as described previously. Biomass data were analyzed similarly, but trapping effort was lower due to loss of body mass information in predated individuals, from which only skulls remained for identification. We also estimated Shannon diversity indices (ZAR 1996) and trapping success for each area.

## RESULTS

The total effective capture effort was 5254 traps.day<sup>-1</sup>. We had 718 successful captures, which included 325 individuals from 14 species of non-flying small mammals. Species included 12 sigmodontine rodents and two marsupials (Tab. I). New morphotypes of *Akodon* Meyen, 1833 and *Scapteromys* Waterhouse, 1837, both with a diploid number of 34 chromosomes – named *Akodon* sp. 2 ( $2n = 34$ ) and *Scapteromys* sp. ( $2n$

Table I. Abundance and diversity of non-flying small mammals in areas protected and impacted by fire and livestock grazing in grassland-Araucaria forest ecotones in southern Brazil, in 2004 (trapping effort of 2327 traps.day<sup>-1</sup> in the protected area and 2927 traps.day<sup>-1</sup> in the impacted area).

Species	Number of individuals		Individuals/100 traps.day <sup>-1</sup>	
	Protected area	Impacted area	Protected area	Impacted area
Didelphimorphia: Didelphidae				
<i>Monodelphis dimidiata</i>	10	–	0.43	–
<i>Philander frenatus</i>	1	–	0.04	–
Rodentia: Cricetidae				
<i>Akodon montensis</i>	46	15	1.98	0.51
<i>Akodon paranaensis</i>	6	–	0.26	–
<i>Akodon</i> sp. 1	1	–	0.04	–
<i>Akodon</i> sp. 2 (2n = 34)	43	–	1.85	–
<i>Brucepattersonius iheringi</i>	2	–	0.09	–
<i>Delomys dorsalis</i>	7	3	0.30	0.10
<i>Oligoryzomys flavescens</i>	3	–	0.13	–
<i>Oligoryzomys nigripes</i>	51	43	2.19	1.47
<i>Oxymycterus nasutus</i>	87	–	3.74	–
<i>Scapteromys</i> sp. (2n = 34)	1	–	0.04	–
<i>Sooretamys angouya</i>	–	1	–	0.03
<i>Thaptomys nigrita</i>	5	–	0.21	–
Total	263	62	11.3	2.12
Shannon index (H')	1.82	0.81		

= 34), respectively) – were captured and remain to be described. We were also unable to identify one captured individual (*Akodon* sp. 1) because it did not match morphological characteristics of any previously described species.

Thirteen species were captured in the protected area and four species were captured in the impacted area. Ten species were found only in the protected area, three in both areas, and *Sooretamys angouya* (Fischer, 1814) was only found in the impacted area (a single individual was captured). The number of captured individuals and the Shannon diversity index were also higher in the protected area (four and two times higher, respectively) (Tab. I). Capture success was five times higher (25.1% to 4.58%) too.

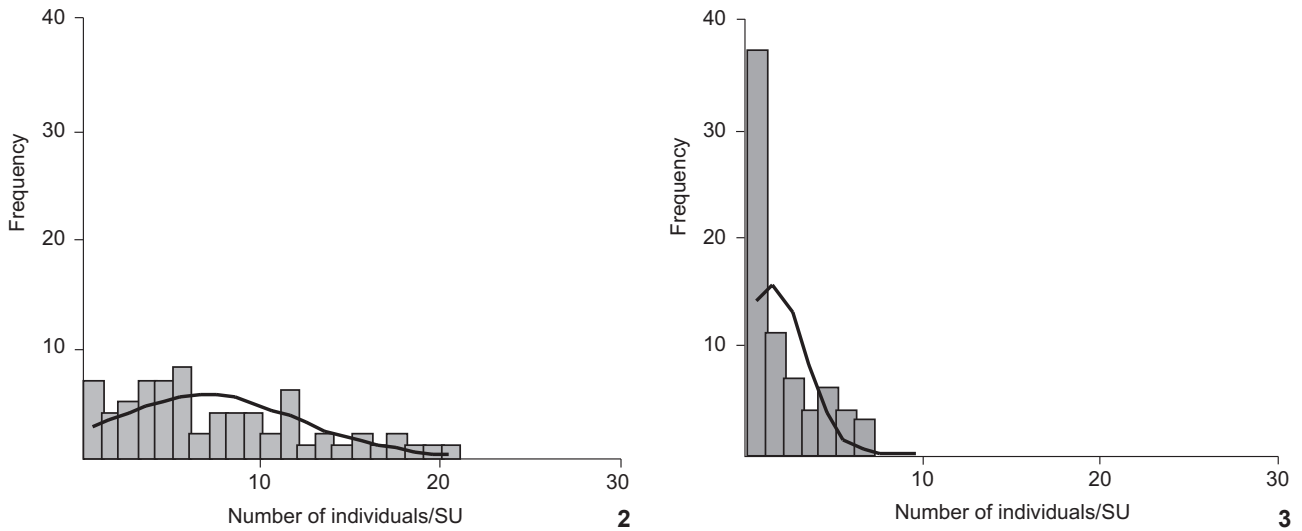
The average number of individuals per sampling unit was 6.75 in the protected area and 1.37 in the impacted area. No captures occurred in more than a half of the sampling units (37, from which 34 were in grasslands) in the impacted area (Figs 2 and 3).

Similarly, the average number of individuals per trap.day<sup>-1</sup> in each sampling unit of the protected area was 6.5 times higher than in the impacted area (Randomization test,  $p = 0.001$ ) (Fig. 4). By applying the same analysis to the biomass data, we found that the average biomass per trap.day<sup>-1</sup> in each

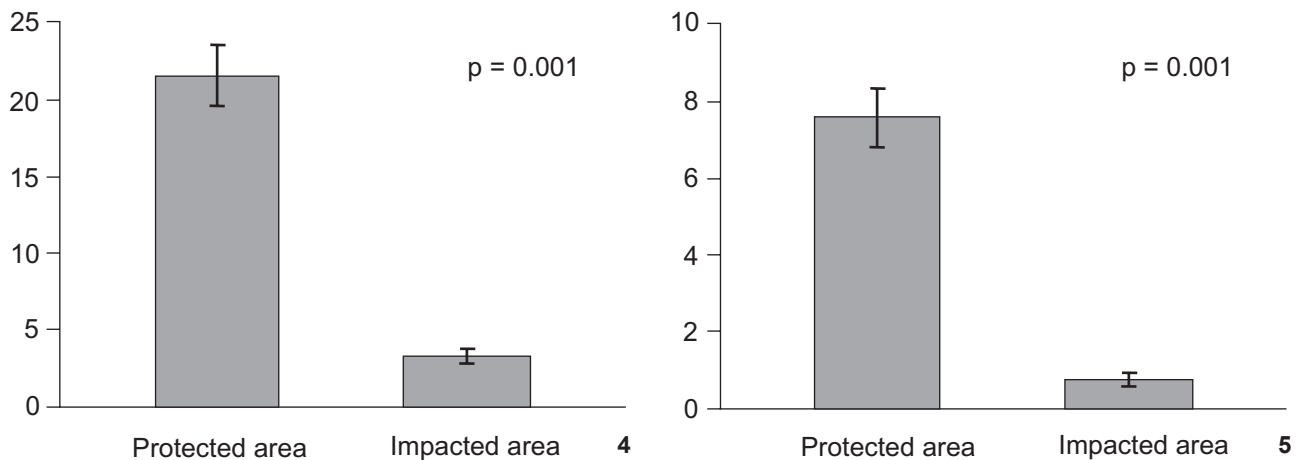
sampling unit was 9.6 times higher in the protected area (Randomization test,  $p = 0.001$ ) (Fig. 5).

The results of the multivariate analysis indicate a significant difference in the composition of non-flying small mammals of the two areas, either when data on abundance of individuals per species (Randomization test,  $p = 0.001$ ) or biomass per species (Randomization test,  $p = 0.001$ ) were used. In these analyses, the effects of factors that might have interfered in the results, such as habitat type (grassland or forest) and distance to borders, were excluded. The results of the test indicated a significant interaction between the factors 'habitat type' ( $p = 0.001$ , either for abundance or biomass) and 'distance to border' ( $p = 0.057$  for abundance and  $p = 0.019$  for biomass) on the analyzed factor: 'management'. Therefore, each habitat type was compared separately, according to management type. By excluding the effect of the factor 'distance to border', we found a significant difference between the protected and impacted areas in grassland ( $p = 0.001$ , either for abundance or biomass) and forest habitats ( $p = 0.001$  for abundance and  $p = 0.003$  for biomass).

Richness and diversity of non-flying small mammals in grassland habitats were higher than in forest habitats (richness: grasslands = 11 species, forests = 10; diversity (Shannon



Figures 2-3. Frequency of number of individuals of non-flying small mammals per sampling unit (trapping station) in (2) protected areas and (3) areas impacted by fire and livestock grazing in grassland-Araucaria forest ecotones in southern Brazil, in 2004 (144 sampling units, a half for each area). The line represents a normal curve.



Figures 4-5. Distribution of mean values (with standard errors) of (4) the number of individuals and (5) biomass (g) per trap.day<sup>-1</sup> in each sampling unit (x100) – trapping effort: 2327 traps.day<sup>-1</sup> in the protected area and 2927 traps.day<sup>-1</sup> in the impacted area; 144 sampling units, a half for each area – in protected areas and in areas impacted by fire and livestock grazing in grassland-Araucaria forest ecotones in southern Brazil, in 2004.

Index): grasslands = 1.619, forests = 1.255). However, only two individuals of *Oligoryzomys nigripes* (Olfers, 1818) were captured in grasslands located in ecotones under the effect of livestock grazing (Fig. 6). In impacted areas located in forest habitats, the abundance of this species was 80% higher than in protected areas (Fig. 6). Differences in species richness and total abundance of individuals remained high, but were lower between both areas in this habitat.

## DISCUSSION

Research on the effects of disturbances such as fire and livestock grazing on non-flying small mammals, encompassing several biomes around the world, has focused on fire experiments (BECK & VOGL 1972, PRICE & WASER 1984, OJEDA 1989, FA & SANCHEZ-CORDERO 1993, VIEIRA & MARINHO-FILHO 1998, FIGUEIREDO & FERNANDEZ 2004). However, research in grassland-



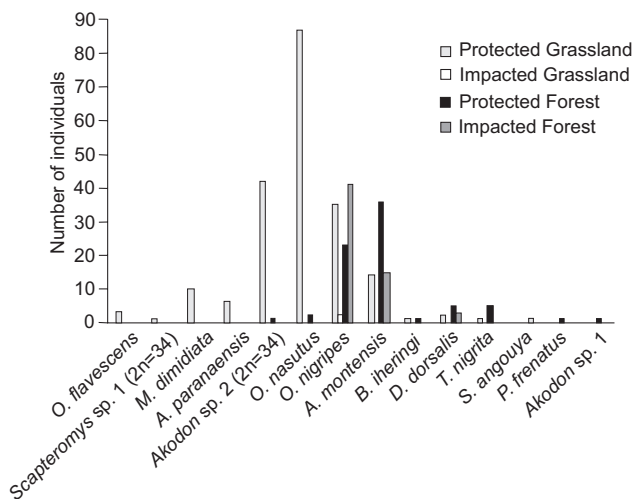


Figure 6. Total number of individuals captured in grassland and forest areas, protected and impacted by fire and livestock grazing in southern Brazil, in 2004 (trapping effort: 1080 traps.day<sup>-1</sup> in protected grassland, 1409 traps.day<sup>-1</sup> in impacted grassland, 1247 traps.day<sup>-1</sup> in protected forest and 1518 traps.day<sup>-1</sup> in impacted grassland).

Araucaria forest ecotones are lacking. This study presents the first analysis in this kind of environment. Moreover, our sampling design is relatively different from other studies, which usually compare the assemblage of non-flying small mammals before and after a disturbance, and monitor the process of recolonization of impacted areas (such as SIMONS (1991), FA & SANCHEZ-CORDERO (1993), VIEIRA & MARINHO-FILHO (1998), VIEIRA (1999) and FIGUEIREDO & FERNANDEZ (2004)). Our findings indicate that richness, abundance, diversity and biomass of non-flying small mammals in grassland-Araucaria forest ecotones subjected to frequent fires and livestock grazing is lower than in areas with the same vegetation type, but without such impacts for at least 10 years.

While forest areas normally suffer indirect and localized direct effects from management, grassland areas are directly impacted. Our findings have shown that species such as *Monodelphis dimidiata* (Wagner, 1847), *Akodon* sp. 2 ( $2n = 34$ ) and *Oxymycterus nasutus* (Waterhouse, 1837), strictly associated with grasslands, are highly abundant in protected areas, but absent in impacted areas with the same kind of original vegetation. The lower spatial heterogeneity inside the impacted grasslands could be a reason for this absence.

*Oligoryzomys nigripes* was the most abundant species in this study, and the less demanding with regard to the degree of habitat conservation (BONVICINO *et al.* 2002). Nevertheless, only two individuals were found in grassland areas subject to frequent fire and livestock grazing. Even though we have not assessed the availability of shelter – which increases predation risk – or food in grasslands subject to frequent fire and grazing,

these seem to be the main factors determining the absence of non-flying small mammal individuals in these impacted grassland habitats.

Some studies point out that fire incidents are followed by drastic declines in non-flying small mammal populations (FA & SANCHEZ-CORDERO 1993). Recovery is relatively faster in sites where fire is frequent (SIMONS 1991, VIEIRA 1999), and where opportunist species are already present in the second month after disturbance (COOK JR 1959, BECK & VOGL 1972, OJEDA 1989, FA & SANCHEZ-CORDERO 1993, VIEIRA & MARINHO-FILHO 1998, VIEIRA 1999, SIMON *et al.* 2002, FIGUEIREDO & FERNANDEZ 2004). This pattern indicates that populations in sites regularly subjected to fire are better adapted to post-fire conditions. Although we have sampled grasslands with no fire incidents for up to 18 months, we have not found any non-flying small mammal species that are able to colonize or re-colonize areas subjected to frequent fires (every one or two years) and livestock grazing. This fact suggests that grazing is the most important factor here.

On the other hand, we have found one species, *O. nigripes*, to be more abundant in impacted forests, indicating its opportunistic character. This finding is consistent with the results of DALMAGRO & VIEIRA (2005), who showed that this rodent is associated with a low density of trees and limited canopy cover in the Araucaria forest. This pattern is attributed to forests with low recruitment of trees, which, in our case, would be an expected consequence of fire and livestock grazing and trampling on the impacted area.

Apparently, our findings differ from those of CARO (2001), carried out in a region dominated by dry deciduous forests in Tanzania. The author found richness and abundance of non-flying small mammals to be higher in areas affected by agriculture and cattle rearing than in a neighboring national park protected from anthropogenic impact. BILENCA *et al.* (2007) also had a high trapping success in a rural landscape, which they attributed to a greater availability of shelter (in cultivated fields), lower predation risk and lower competition with ungulates. In fact, these factors also seem to drive the dynamics of the species in non-flying small mammals, because in our study the greatest values of richness, abundance, diversity and biomass were found in protected areas. These protected areas presented greater cover and biomass in both tree and herb layers, a source of food and shelter from predators and harsh climatic conditions, only available in the absence of livestock.

Finally, the record of at least two new species (associated with grasslands) not yet described by science makes evident the urgent need for more studies in Araucaria forest ecotones. For instance, studies that correlate the assemblage of non-flying small mammals with microhabitat characteristics in grassland-Araucaria forest ecotones are very important. Moreover, the use and frequency of fire, and traditional livestock management practices, currently extensive and practiced with no restrictions to the use of forest patches, should be reviewed and minimized in order to ensure the preservation of the di-

iversity of non-flying mammals in grasslands-Araucaria forests ecotones. This study stresses the importance of conserving the forest-grassland mosaic of the south Brazilian highlands.

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