

Effects of fire and cattle grazing on amphibians and lizards in northeastern Argentina (Humid Chaco)

Paula D. Cano · Gerardo C. Leynaud

Received: 25 February 2009 / Revised: 21 September 2009 / Accepted: 1 October 2009
© Springer-Verlag 2009

Abstract Richness, diversity and abundance of amphibians and lizards were compared in four sites subjected to different fire and grazing regimes in Mburucuyá National Park (Corrientes, Argentina). The surveys were performed using drift fence–pitfall traps randomly distributed in four sites subjected to the following prescribed fire and grazing regimes: (A) no fire or grazing for the last 12 years (control), (B) 3 years since the last prescribed fire, (C) annual prescribed fire and (D) annual prescribed fire and grazing. Fourteen species of amphibians and six species of lizards were captured during the study period (80 days). The lowest values of amphibian richness and abundance were found in sites C and D. The amphibian species *Leptodactylus ocellatus* and *Leptodactylus podicipinus* were most abundant in the control site (A). In lizards, no significant differences were found among sites, although *Kentropyx viridistriga* and *Teius oculatus* were more abundant in annually burnt sites (C and D). Vegetation variables were correlated with amphibian and lizard species composition. The species differed in their preference for the differently managed sites. The combination of fire and grazing had a negative effect on diversity of amphibians

and lizards, with a greater effect on amphibians. Also, fire favoured the presence of some lizard species.

Keywords Prescribed fire · Disturbance · Grasslands · Herpetofauna · Cattle · Argentina

Introduction

In several regions of Argentina, fire and grazing are important disturbances that critically modify the landscape. In wild areas, fire reduces plant biomass, suppresses development of woody vegetation and removes the upper litter layer (Gibson and Hulbert 1987; Ensbey et al. 2000). At the same time, fire exposes soil where new individuals can establish, leading to different successional stages of plant and animal communities (Kunst et al. 2003). Grazing by domestic cattle has a significant effect on grasslands through plant removal and compaction, erosion and loss of soil biological properties (Braunack and Walker 1985; Abril and Bucher 1999; Ensbey et al. 2000). Grazing-induced changes modify availability of habitat and food resources for the grassland fauna (Ryder 1980).

Some species of amphibians and reptiles are highly sensitive to environmental changes induced by fire and grazing. Some particular characteristics of amphibians (highly permeable skin, shell-less eggs) make them vulnerable to drastic changes in environmental quality (Zug et al. 2001). Reptile species are more tolerant to on-site moisture reductions than amphibian species mainly because reptiles' skin is relatively impermeable, their eggs have a shell and also because some aspects of their behaviour (use of borrows and low mobility) help them overcome adverse conditions. However, as reptiles are associated with particular biomes, some species are highly sensitive to

Communicated by H. Kierdorf

P. D. Cano
Programa de Maestría en Manejo de Vida Silvestre,
Centro de Zoología Aplicada, Facultad de Ciencias Exactas,
Físicas y Naturales, Universidad Nacional de Córdoba,
Casilla de Correos 122,
5000 Córdoba, Argentina

G. C. Leynaud (✉)
Centro de Zoología Aplicada, Facultad de Ciencias Exactas,
Físicas y Naturales, Universidad Nacional de Córdoba,
Casilla de Correos 122,
5000 Córdoba, Argentina
e-mail: gleynaud@efn.uncor.edu

subtle changes in the ecological communities where they belong (Zug et al. 2001). A species' local distribution and abundance may be restricted by fire-induced changes (James and M'Closkey 2003). Pianka (1996) mentioned fire as one of the most important factors determining lizard species diversity in the deserts of Australia.

Mortality of individuals is one of the immediate effects of fire on animal populations (Heinrich and Kaufman 1985; Whelan 1995). However, different authors mentioned that neither amphibians nor reptiles are usually directly affected by fire or that they are only slightly affected (Mushinsky and Gibson 1991; Cunningham et al. 2002). Accordingly, studies conducted in other parts of the world indicate that fire has a low effect on amphibians and reptiles (Cunningham et al. 2002). Many species even take advantage of the habitat mosaic generated by fire (Mushinsky and Gibson 1991) and of the increased availability of food and refuge, the latter one being provided by logs (Cunningham et al. 2002; James and M'Closkey 2003). Furthermore, the herpetofauna historically adapted to environments with periodic natural fire regimes not only tolerates but also takes advantage of this disturbance (Russell et al. 1999; Brisson et al. 2003).

Prescribed burning, usually conducted during winter and early spring, is a management tool used to reduce hazardous fuel buildup, maintain or restore native ecosystems, increase water availability, control undesirable plant species and create clean seedbeds for seeding purposes (Wright and Bailey 1982). Prescribed fire has also been indicated as an adequate management action to improve pasture quality (Andersen et al. 2005) and restore historical mosaics of successional stages, habitat structure and plant species composition (Russell et al. 1999). However, most authors suggest that the effects of fire on biodiversity are complex, depend on the environment and species analysed and pose an important challenge to wildlife managers (Mushinsky 1985; Penn et al. 2003).

Grasslands in northeastern Argentina are heavily exploited for cattle production, the main economic activity in the region; fire is used as a management tool to promote re-sprouting and remove barely palatable vegetation (Whelan 1995). One of the main problems encountered in the study of the effect of fire or overgrazing on animal communities or populations is the low availability of undisturbed sites relatively close to sites presenting a mosaic of disturbances induced by fire and grazing. This study aims at examining the response of amphibians and lizards to the different management strategies implemented in an area that presents a disturbance mosaic generated and maintained for experimental purposes by means of prescribed burning and cattle grazing.

The study was conducted in Mburucuyá National Park, one of the largest native grasslands of northeastern

Argentina, which provides a unique opportunity to evaluate the effects of fire and grazing, individually and combined, on local populations of amphibians and lizards.

Materials and methods

Study area

The work was conducted in Mburucuyá National Park (17,682 ha; 57°59' and 58°08' W; 27°58' and 28°05' S), located in the northwest of the province of Corrientes, northeastern Argentina (Fig. 1). The relief in the region is slightly undulated, with deficient drainage due to the slight slope; 65% of the area is covered by semi-permanent ponds, swamps and marshes (Saibene and Montanelli 1997).

Mean annual temperature ranges between 21°C and 23°C. Maximum absolute temperatures can peak 46.5°C, and minimum absolute values of up to -5.5°C are recorded in winter. Frosts are barely frequent, with up to 340 days free of frosts a year. Mean annual rainfall is 1,300 mm, concentrated in spring and summer, with a dry season in winter.

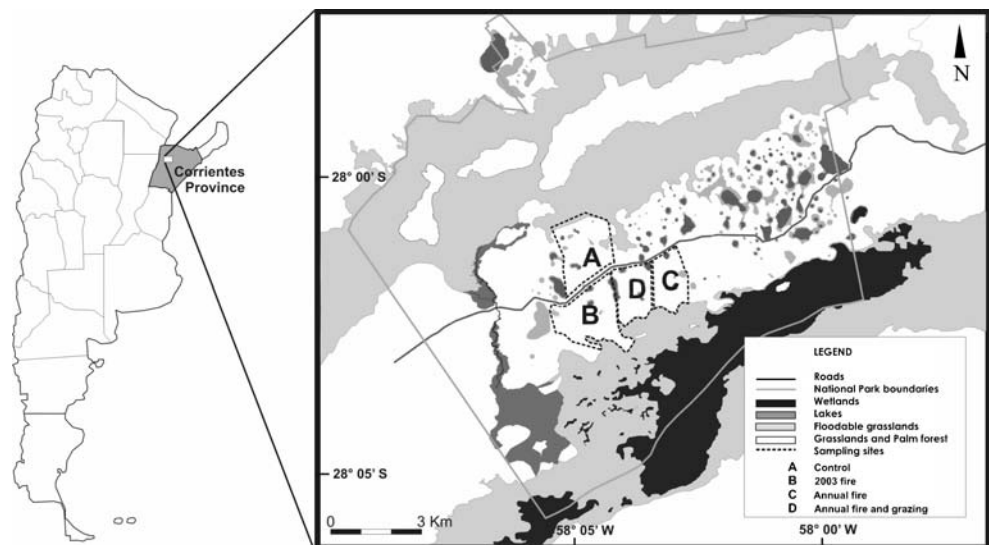
The park lies in a wide ecotone among the Paranaense, Chaco and Espinal phytogeographic provinces (Cabrera 1976). The area is included in the Ibera wetlands system, which is considered one of the most important wetlands in Latin America (Canevari et al. 1998). The characteristic physiognomy of the Park is of an open savannah, with grasslands and inclusion of hygrophilous forests. The most common species of grasses during the study period were *Andropogon lateralis*, *Andropogon selloanus*, *Schizachyrium paniculatum*, *Sorghastrum agrostoides*, *Elionurus muticus* and *Paspalum notatum*.

Experimental design

Four sampling sites (minimum of 400 ha each) with different history of grazing and natural or prescribed fires were selected. The sites were defined as (A) site non-burned and non-grazed since 1995, (B) site non-burned since 2003 and non-grazed, (C) site annually burned and non-grazed and (D) site annually burned and grazed (Fig. 1).

Burns were conducted at the time of the year of greatest accumulation of dry vegetation cover (August–September). Fuel material was mostly fine (dried grasses), with an approximate load of 5 t/ha and about 10–15% humidity. Head fires were ignited with a hand-held drip torch. Fires reached a mean height of 3 to 5 m, with a maximum of 10 m. Prescribed burns were conducted under the following conditions: winds 10–15 kph, air temperature 15–22°C and relative humidity 40–50%.

Fig. 1 Location of Mburucuyá National Park (Corrientes, Argentina) and of the four sampling sites (A, B, C and D)



Cattle carrying capacity in the grazed sites was 3.1 ha per cattle unit. Cattle used the plots all the year round. Five to 7 days before the start of the start of burns, they were removed and were re-introduced in the burned plots 2 months after the last burn.

The four sites were simultaneously sampled to compare richness, abundance and diversity of amphibians and lizards and the relationship of these parameters with vegetation structure and composition in each site. Samplings were conducted uninterruptedly from January 12 to April 1, 2006 (80 days). Samplings taken in the grazed plots were conducted when cattle were present.

Because of the experimental design used, it was not practical to find other areas that replicate the landscape treatments because natural grasslands are very scarce in this portion of the Chaco. Indeed, Mburucuyá National Park is the largest relict that has been barely modified by man. Therefore, we will restrict our statistical inferences to these particular areas, considering our sampling methodology as adequate to represent these them. In many cases, it is difficult to replicate or randomise appropriately perhaps because lack of enough wild areas to be included in the analysis (as this work) or financial limitations; however, as stated by Wester (1992), the fact that appropriate replication or randomisation is not adequately incorporated into a study does not mean that the study lacks useful information.

Samplings of amphibians and lizards were conducted using drift fence–pitfall traps. Trapping arrays consisted of three drift fence arms radiating out at 120° from a central point, with three buckets at the outer edge of every drift fence and one bucket placed at the centre of the array (Greenberg et al. 1994). Drift fences 8 m in length and 50 cm in height were constructed with polypropylene shade cloth arranged in a straight line. The 20-L buckets were buried in the ground, with the opening flush with the

surface; the buckets were filled with a wet sponge, 1–2 cm of water, and plant remains to prevent dehydration of individuals (Greenberg et al. 1994). Thus, each array was a sampling unit made up of three drift fences and four buckets (Leynaud and Bucher 2005).

The sampling statistical design used is a completely randomised model with one classification factor (Zar 1984). In each of the four sampling sites (A, B, C and D), eight sampling units were randomly established. A total of 32 arrays (sampling units) were installed in the entire study area at a minimum distance of at least 500 m from one another. Traps were checked once or twice a day. Captured animals were marked by toe clipping to avoid double counting (Masters 1996; Jones et al. 2000); specimens were photographed and released at the point of capture (Jones 1988).

Data analysis

Richness (number of species) and relative abundance (number of individuals captured) were estimated per group (amphibians and lizards) and per species in the four sites. Richness and abundance were compared among sites with an analysis of variance (ANOVA). Normal distribution of records and homogeneity of their variances were evaluated using the Shapiro–Wilk test and the Levene test, respectively; both tests were applied on residuals and absolute residuals of the variables analysed (Zar 1984). Where data sets were not normally distributed, a Kruskal–Wallis one-way test was executed, followed by Dunn’s method of multiple comparisons. Diversity of amphibians and reptiles was estimated with the Shannon index. Significant differences in the index value among sites were evaluated using the *t* test (following Magurran 1988). Similarity of assemblage composition among sites was evaluated using Sorensen’s similarity index.

Description of vegetation and its relationship with amphibian and reptile fauna

Plant species composition is described in the sampling sites, and the following plant and structural habitat variables in each array were measured: (a) ground cover (bare, litter and/or grass), (b) grassland height, (c) litter depth, (d) richness and diversity of plant species and (e) distance to forest patches. To do this, 30 points were randomly defined around each trap. Plant species were recorded at each point. Ground cover (bare, litter and/or grass) was measured with an iron frame (20×50 cm) placed at each of the points to estimate percent cover. Grassland height and litter depth were measured with a graded rod at each point, obtaining a mean value (centimetres) for each trap (Sutherland 1996). Richness and diversity of plant species were estimated by counting species in 30 sampling quadrats (5×5 m), located in each array. Species richness was expressed as the number of species (average and total) in each site, whereas diversity was expressed with the Shannon index. The minimum distance (metres) from each sampling unit to the nearest forest patch was measured. Data were analysed using ANOVA, and means were separated using Tukey's test ($P < 0.05$). An ANOVA test was also conducted on arcsine-transformed percentages.

To describe the association between plant composition and physiognomic characteristics or between structure and composition and abundance of amphibians and lizards in each site, a canonical correspondence analysis (CCA) was performed (Jongman et al. 1995), using PC-ORD software (Mc Cune and Grace 2002). This test estimates correlations between multiple variables and evaluates their statistical significance (Mc Garigal et al. 2000; Mc Cune and Grace 2002); it also separates the species of interest and generates ordination gradients. In this work, analyses with two data sets were performed. First, amphibian and lizard species composition was analysed relative to plant species composition in each site. Second, amphibian and lizard species composition was analysed relative to the vegetation structure variables of each site. The statistical significance of the results was determined with the Monte Carlo test using PC-ORD ($P < 0.05$).

Results

Effect of the different grassland management systems on amphibians and lizards

A total of 135 amphibians (14 species, five families) and 106 lizards (six species, five families) were captured throughout the 80-day sampling period. Individual species

captures are given in Tables 1 and 2. Results of the comparison of amphibian and lizard assemblages among sites are expressed separately.

Amphibians Assemblages in the four sites (A, B, C and D), compared using ANOVA with a posteriori test, differed significantly in richness ($F=5.28$, $P=0.006$) and abundance ($F=3.25$, $P=0.039$). The highest diversity values (Shannon index) were in A, B and C sites, and the lowest diversity value was recorded in site D (Table 3). The highest richness and abundance values were found in sites A and B, whereas the lowest ones were recorded in sites C and D (Table 3).

Three species (*Leptodactylus diptyx*, *Leptodactylus gracilis* and *Scinax nasicus*) were recorded only in site A, and one species (*Physalaemus riograndensis*) was present only in site B. The species *Elachistocleis bicolor*, *Leptodactylus ocellatus*, *Leptodactylus podicipinus* and *Physalaemus albonotatus* were recorded in all sites (A, B, C and D).

Only two amphibian species exhibited significant differences among sites (Table 1): *L. ocellatus* was more abundant in sites A and B; *L. podicipinus* was more abundant in sites A and D. The highest similarity in species composition was found between sites B and C (Sorensen's similarity index=0.84), and the lowest similarity was found between sites A and D (0.58). A high value was observed between sites A (control) and site B (0.63).

Lizards Assemblages of the different sites (A, B, C and D), compared using ANOVA with a posteriori test, were not significantly different in richness ($F=2.16$, $P=0.11$) and abundance ($F=1.86$, $P=0.16$). The values of diversity (Shannon index) were similar among the four sites (Table 4). *Tupinambis merianae* was found only in site A, *Ophiodes intermedius* was recorded only in site B, *Kentropyx viridistriga* was recorded only in sites C and D and *Mabuya dorsivittata* and *Cercosaura schreibersii* were recorded in all sites (A, B, C and D). The species *K. viridistriga* exhibited the greatest abundance in site C and was not found in sites A and B, whereas the species *M. dorsivittata* exhibited the greatest abundance values in sites A and B. *Teius oculatus* showed the highest values in site C. The highest Sorensen's similarity index value was found between sites C and D (0.97).

Relationship between amphibian and lizard fauna and vegetation

Plant composition (richness and diversity) did not differ significantly among sites ($F=2.57$, $P > 0.05$ and $F=2.37$, $P > 0.05$, respectively). Regarding variables describing vegetation structure and physiognomy, grassland height

Table 1 Abundance of individuals (n =abundance; $\bar{x}\pm 1$ SE) of each amphibian species in the four sampling sites

Amphibians species	Site A			Site B			Site C			Site D		
	n	\bar{x}	\pm SE	n	\bar{x}	\pm SE	n	\bar{x}	\pm SE	n	\bar{x}	\pm SE
<i>Leptodactylus diptyx</i>	1	0.13	\pm 0.13	0	0	0	0	0	0	0	0	0
<i>Rhinella schneideri</i>	1	0.13	\pm 0.13	6	0.75	\pm 0.41	0	0	0	1	0.13	\pm 0.1
<i>Elachistocleis bicolor</i>	2	0.25	\pm 0.25	1	0.13	\pm 0.13	2	0.25	\pm 0.16	3	0.38	\pm 0.3
<i>Leptodactylus gracilis</i>	1	0.13	\pm 0.13	0	0	0	0	0	0	0	0	0
<i>Leptodactylus ocellatus</i>	11	1.38 a	\pm 0.46	10	1.25 a	\pm 0.62	1	0.13 b	\pm 0.13	2	0.25 b	\pm 0.3
<i>Leptodactylus podicipinus</i>	16	2 a	\pm 0.63	2	0.25 b	\pm 0.16	4	0.5 b	\pm 0.19	14	1.75 a	\pm 1.3
<i>Lysapsus limellus</i>	5	0.63	\pm 0.26	3	0.38	\pm 0.18	2	0.25	\pm 0.25	0	0	0
<i>Odontophrynus americanus</i>	0	0	\pm 0	0	0	0	1	0.13	\pm 0.13	1	0.13	\pm 0.1
<i>Physalaemus albonotatus</i>	8	1	\pm 0.73	9	1.13	\pm 0.35	2	0.25	\pm 0.16	1	0.13	\pm 0.1
<i>Physalaemus biligonigerus</i>	0	0	\pm 0	1	0.13	\pm 0.13	2	0.25	\pm 0.16	1	0.13	\pm 0.1
<i>Physalaemus riograndensis</i>	0	0	\pm 0	1	0.13	\pm 0.13	0	0	0	0	0	0
<i>Physalaemus santafecinus</i>	0	0	\pm 0	1	0.13	\pm 0.13	2	0.25	\pm 0.16	0	0	0
<i>Pseudopaludicola mystacalis</i>	3	0.38	\pm 0.26	12	1.5	\pm 0.87	1	0.13	\pm 0.13	0	0	0
<i>Scinax nasicus</i>	1	0.13	\pm 0.13	0	0	0	0	0	0	0	0	0

Species indicated in boldface exhibited significant differences (Kruskal–Wallis, $P\leq 0.05$). Different letters indicate significant differences among sites (Dunn’s test)

($F=46.49$, $P<0.02$), litter depth ($F=7.89$, $P<0.04$), ground cover ($F=4.66$, $P<0.01$) and distance to the nearest forest patch ($F=3.52$, $P<0.02$) were significantly different among sites (Table 5). Site A (control) had the highest values of grassland height and litter depth, and the lowest value of ground cover was recorded in site C (Table 5). Results of CCA are expressed separately for amphibians and reptiles.

Amphibians CCA between amphibian assemblage composition and plant species composition was not statistically significant ($P>0.05$). Accordingly, plant species composition in each site cannot account for the differences detected in amphibian assemblage composition. Moreover, CCA correlations of amphibian assemblage and vegetation

structure variables were statistically significant ($P<0.05$). The first three axes accounted for 35% of the variability observed. In the ordination diagram, a separation of the sampling units (belonging to the same treatments) relative to grassland management is observed (Fig. 2). Axis 1 was positively correlated with the variables grassland height ($r=0.72$, $P<0.05$), litter depth, ($r=0.65$, $P<0.05$) and diversity ($r=0.78$, $P<0.05$) and was inversely correlated with plant species richness ($r=-0.61$, $P<0.05$). Sampling units corresponding to the annually burned sites (C and D) were to the left of axis 1, whereas sites with the longest time since last burn (A and B) were to the right (Fig. 2). The species *Physalaemus santafecinus*, *Physalaemus biligonigerus*, *E. bicolor*, *L. podicipinus* and *Odontophrynus americanus* (to the left of axis 1) were correlated with

Table 2 Abundance of individuals (n =abundance; $\bar{x}\pm 1$ SE) of each lizard species in the four sampling sites

Lizard species	Site A			Site B			Site C			Site D		
	n	\bar{x}	\pm SE	n	\bar{x}	\pm SE	n	\bar{x}	\pm SE	n	\bar{x}	\pm SE
<i>Kentropyx viridistriga</i>	0	0 b	\pm 0	0	0 b	\pm 0	6	0.75 a	\pm 0.4	1	0.13 b	\pm 0.13
<i>Mabuya dorsivittata</i>	12	1.5 b	\pm 0.6	16	2 b	\pm 0.5	2	0.25 a	\pm 0.2	3	0.38 a	\pm 0.26
<i>Ophiodes intermedius</i>	0	0	\pm 0	2	0.25	\pm 0.3	0	0	\pm 0	0	0	\pm 0
<i>Cercosaura schreibersii</i>	6	0.75	\pm 0.3	4	0.5	\pm 0.4	6	0.75	\pm 0.3	3	0.38	\pm 0.26
<i>Teius oculatus</i>	1	0.13 b	\pm 0.1	0	0 b	\pm 0	30	3.75 a	\pm 1	10	1.25 b	\pm 0.45
<i>Tupinambis merianae</i>	4	0.5 b	\pm 0.3	0	0 a	\pm 0	0	0 a	\pm 0	0	0 a	\pm 0

Species indicated in boldface exhibited significant differences (Kruskal–Wallis, $P\leq 0.05$). Different letters indicate significant differences among treatments (Dunn’s test)

Table 3 Variables of amphibian diversity in sites A, B, C and D

	Site A	Site B	Site C	Site D
Richness	10(\bar{x} = 3.38) a	10(\bar{x} = 3.25) a	9(\bar{x} = 2.00) b	7(\bar{x} = 1.38) b
Abundance	49(\bar{x} = 6.13) a	46(\bar{x} = 5.75) a	17(\bar{x} = 2.13) b	23(\bar{x} = 2.88) b
Shannon diversity	1.85 a	1.91 a	2.10 a	1.33 b

Richness and abundance were compared using ANOVA and Tukey's test. Shannon indices were compared in pairs, using *t* test (Magurran 1988). Different letters indicate significant differences ($P \leq 0.05$)

the variable plant species richness and were inversely correlated with grassland height, litter depth and diversity. The species *S. nasicus*, *L. diptyx*, *L. gracilis*, *P. albonotatus*, *Lysapsus limellus*, *Pseudopaludicola mystacalis*, *L. ocellatus*, *P. riograndensis* and *Rhinella schneideri*, located to the right of axis 1, were positively correlated with grassland height, litter depth and diversity and were inversely correlated with richness.

Lizards CCA between lizard assemblage composition in each site and plant species composition was not statistically significant ($P > 0.05$). Accordingly, plant species composition in each site cannot account for the differences detected in lizard assemblage composition. CCA correlations between lizard assemblages and vegetation structure variables were statistically significant ($P < 0.05$). The first three axes accounted for 40% of the variability. Axis 1 was strongly correlated with variables grassland height ($r = 0.943$, $P < 0.05$), litter depth ($r = 0.706$, $P < 0.05$) and diversity ($r = 0.789$, $P < 0.05$) and inversely correlated with plant species richness ($r = -0.789$, $P < 0.05$). As we observed for amphibians, in the ordination diagram, axis 1 separated sampling units relative to grassland management: Sampling units belonging to sites C and D were to the left, and sampling units corresponding to sites with the longest time since last burn (A and B) were to the right (Fig. 3). The species *K. viridistriga*, *T. oculatus* and *C. schreibersii* were positively correlated with richness and negatively correlated with grassland height, litter depth and diversity. The species *O. intermedius*, *T. merianae* and *M. dorsivittata*, located to the right of axis 1, were positively correlated with variables grassland height, litter depth and diversity.

Discussion

Most of the studies on the effect of fire on amphibians and lizards are generally conducted in the short and mid term (Minshall et al. 1997; Russell et al. 1999; Radke et al. 2008). Some studies indicate that the immediate effect of fire on an assemblage of amphibians and lizards is a remarkable reduction of overall diversity (McLeod and Gates 1998; Schurbon and Fauth 2003). However, it is interesting to analyse how the disturbed environment evolves over a longer period. The present study, which evaluates grassland management with different burning time since last fire and presence of cattle in similar environments during 12 years, shows the extent to which an amphibian and lizard assemblage is affected by these disturbances. The present results also show the changes of the effect of fire on the assemblage in the short (less than 1 year), mid (1–10 years) and long term (more than 10 years), as Minshall et al. (1997) recommend.

At the landscape scale, the fact that total diversity (beta diversity) in the four sites (A, B, C and D) has been greater than in each site separately suggests that heterogeneity driven by fire and grazing favours a greater diversity, which agrees with findings in the Monte region in Argentina (Marone 1990). We agree with other authors (Masters 1996; McLeod and Gates 1998; Jones et al. 2000; Andersen et al. 2005; Wilgers and Horne 2006) in that mosaics of burned/non-burned sites maximise amphibian and lizard diversity because some species prefer open areas with no vegetation whereas other species prefer sites with abundant vegetation and soils with abundant moisture and litter.

Amphibian abundance, richness and diversity increase with longer time since last burn. These parameters had

Table 4 Variables of lizard diversity in sites A, B, C and D

	Site A	Site B	Site C	Site D
Richness	4(\bar{x} = 1.50) a	3(\bar{x} = 1.38) a	4(\bar{x} = 2.38) a	4(\bar{x} = 1.50) a
Abundance	23(\bar{x} = 1.12) a	22(\bar{x} = 1.14) a	44(\bar{x} = 1.72) a	17(\bar{x} = 1.05) a
Diversity Shannon index	1.13 a	0.76 a	0.95 a	1.09 a

Richness and abundance were compared using ANOVA and Tukey's test. Shannon indices were compared in pairs, using *t* test (Magurran 1988). Different letters indicate significant differences ($P \leq 0.05$)

Table 5 Variables of vegetation structure, richness and diversity of plant species in the sampling sites

	Site A		Site B		Site C		Site D	
	\bar{x}	$\pm SE$	\bar{x}	$\pm SE$	\bar{x}	$\pm SE$	\bar{x}	$\pm SE$
Species richness	3.25 a	± 0.62	2.75 a	± 0.53	4.63 a	± 0.42	3.50 a	± 0.38
Diversity (Shannon index)	0.61 a	± 0.09	0.63 a	± 0.08	0.37 a	± 0.05	0.57 a	± 0.08
Grassland height (cm)	158 a	± 0.06	137 b	± 0.06	77 c	± 0.1	57 c	± 0.05
Litter depth (cm)	38 a	± 0.06	26 a	± 0.04	1.3 b	± 0.09	1 b	± 0.01
Ground cover (%)	100 a	0	100 a	0	94.4 b	± 2.54	89.9 b	± 0.1
Distance to forest (m)	61.2 a	± 16.4	173 b	± 52.6	91.2 a	± 12.6	70 a	± 12.8

Variables in boldface exhibited significant differences with ANOVA ($P \leq 0.05$). Different letters indicate significant differences between sites (Tukey's test)

higher values in sites with and without presence of cattle (sites A and B). However, the close similarity of amphibian composition and diversity in sites A and B (Sorensen's similarity index 0.63) shows that in a short period (3 years), a grassland subjected to fire is close to the value in the control treatment (site A), at least relative to the establishment or re-establishment of an amphibian assemblage. Therefore, we may conclude that fire has a short-term

negative effect on amphibian populations, but after a minimum period of 3 years without fire, the environment recovers and effects do not persist. We cannot draw such definite conclusions for overgrazing because of the similarity found in parameters between treatments C and D. However, site D had the lowest diversity values.

No similar studies have been conducted in this portion of Humid Chaco in Argentina. Our study can be compared, for example, with the work by Jones et al. (2000), who also observed that richness and abundance of amphibians are greater with longer time elapsed after a fire. Similar results have been reported by Moseley et al. (2003), who also indicated that amphibian abundance is negatively correlated with the percentage of bare soil. A lower abundance of amphibians in burned sites was also found in forest environments (McLeod and Gates 1998; Schurbon and Fauth 2003).

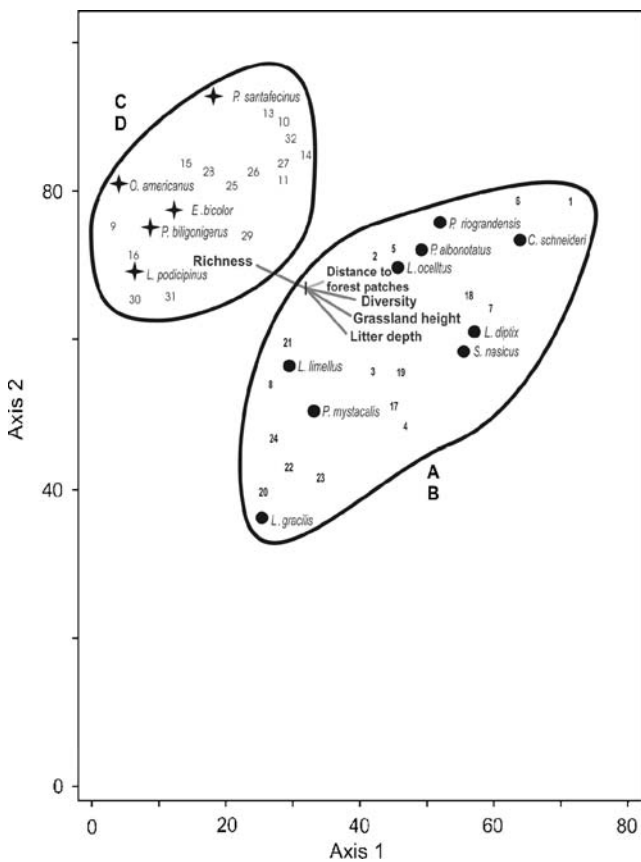


Fig. 2 Canonical correspondence analysis (amphibian assemblage composition and vegetation structure). The locations of amphibian species and sites along the axes are indicated (1–8 2003 fire, 9–16 annual fire with grazing, 17–24 control, 25–32 annual fire)

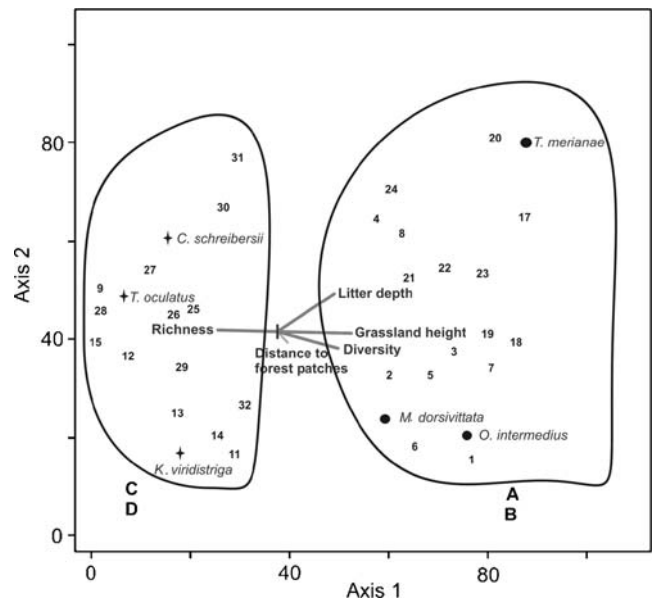


Fig. 3 Canonical correspondence analysis (lizard assemblage composition and vegetation structure). The location of lizard species and sites along the axes are indicated (1–8 2003 fire, 9–16 annual fire with grazing, 17–24 control, 25–32 annual fire)

Significant differences in species composition in assemblages found in areas with different fire and grazing regimes are closely related to the characteristics of the vegetation structure. The loss of plant material deposited on the ground results in reduced soil moisture, which negatively affects the dermal respiratory capacity of amphibians (Duellman and Trueb 1994). This has been mentioned as the main factor affecting abundance and diversity of amphibians in an area (Ash 1995, 1997). Among the requirements of amphibians in terrestrial environments, refuge availability to avoid desiccation is crucial, which explains the absence of many species in disturbed areas with evident absence of litter (Moseley et al. 2003; Bury 2004). Thus, some species, such as *L. ocellatus* and, to a lesser extent, *Scinax nasicus*, *L. diptyx*, *L. gracilis*, *L. limellus*, *P. mystacalis*, *P. albonotatus*, *P. riograndensis* and *Rhinella schneideri*, show a close association with areas completely covered by grasses, with a height of above 1 m and a thick, deep litter layer; these characteristics have been observed in the sites where fire had occurred more than 3 years before (A and B). Furthermore, the sites with more than 3 years since the last fire (C and D), where small areas of bare soil and low grasses occur, are rapidly colonised by species adapted to tolerate dry conditions and to be buried during a long part of the year, such as *O. americanus*, species of nocturnal habits like *E. bicolor* or species like *L. podicipinus*, *P. biligonigerus* and *P. santafecinus*, with a rapid larval development that use temporary ponds for that purpose. These species have a distribution range that show their adaptations to great environmental changes, like those occurring in the Humid Chaco, and their capacity to tolerate arid conditions (Cei 1980; Gallardo 1987).

Unlike observations on amphibians, the different fire and grazing regimes do not affect richness, abundance or diversity in lizards at the level of order. These results agree with works conducted in grasslands in Kansas (Wilgers and Horne 2006) and in forest environments (Cunningham et al. 2002; Fredericksen and Fredericksen 2002; Moseley et al. 2003), where, as we indicated, abundance of some lizard species increases in areas subjected to a disturbance, such as fire. This observation has also been made by Vitt et al. (1998) for another type of disturbance: the extraction of trees in the Amazon forest. In the present work, the only differences observed are related to species composition of the assemblages in each of the landscapes resulted from both disturbances, where few species increase or decrease significantly in abundance in areas subjected to fire or cattle production. Accordingly, a significantly greater number of *K. viridistriga* and *T. oculatus* were captured in sites C and D than in the remaining sites, which shows a preference for open habitats, reduced litter layer and low grasslands. Similar results were found by Pelegrin et al. (2009) in *Teius teius* in an area of the semiarid Chaco. According to the

literature, the species *K. viridistriga* and *T. oculatus* prefer open habitats and margins of bodies of water (Achaval and Olmos 2003; Scrocchi and Giraudo 2005), which would indicate a preference for habitats that have no cover as a consequence of fire. An increase in the percentage of bare soil in burned sites results in an increase of the surface area necessary for the thermoregulation activity of reptiles (Moseley et al. 2003).

By contrast, the species *M. dorsivittata* and *T. merianae* were more abundant in sites where the disturbance occurred earlier (A and B). These species are diurnal, terrestrial and prefer habitats with dense vegetation where to hide and protect from high summer temperatures. They are found in forests and wetlands or on lake margins; they hide in the litter and the vegetation or submerge in the water where they can swim very well (Colli et al. 2002; Achaval and Olmos 2003; Scrocchi and Giraudo 2005). The number of *T. merianae* individuals captured, however, is not enough to draw definite conclusions, but only allows us to indicate a tendency that might be confirmed in future works. Therefore, these species might benefit from habitats that are not frequently burned and that preserve a deep litter layer. Previous studies of the effect of fire on lizards provide similar results and report on the colonisation of burned areas where the species were not present before and on an increase in diversity and abundance of lizards after prescribed fires (Means and Campbell 1981; Mushinsky 1985; Brisson et al. 2003). The increase is attributed to a greater opportunity for thermoregulation found in burned sites (Moseley et al. 2003).

Furthermore, abundance of *K. viridistriga* and *T. oculatus* was negatively affected by the presence of cattle in the annually burned site (site D) with respect to the site that is also annually burned and is not subjected to grazing. Similar results were found by Jones (1981), who documented that grazing reduces abundance and diversity of lizards in different habitats, affecting especially the populations of species of broad habitat.

Responses to changes in vegetation may differ among species, especially considering their preferences for forests or open vegetation patches, the two principal habitat types that make up the landscape of savannas. In this study, the observed changes in the vegetation may affect habitat selection by lizards by increasing temperature, which may influence lizards' thermoregulatory behaviour (Vitt et al. 2007), and by changing microhabitat availability (Vega et al. 2000). For example, while heliothermal lizards (*K. viridistriga*, *T. oculatus* and *C. schreibersii*) show preferences for open habitats (Cei 1993) because of the high solar radiation, other lizards with preference for habitats with forest cover (*O. intermedius*, *T. merianae* and *M. dorsivittata*) might require high moisture and greater shade (Vitt et al. 1998). Thus, our results confirm the

importance of gradients of heterogeneous habitats for lizard diversity.

We conclude that fire plays an important role in the dynamics of grasslands, with a marked influence on diversity and structure of amphibian and lizard fauna.

Acknowledgements We are grateful to Administración de Parques Nacionales for permission to work in Mburucuyá National Park and for providing logistic support. We thank all those who helped in field sampling: H. Ball, A. Vallejos, E. Drewniak, L. Oliva, N. Acevedo, J. Sotelo and E. Periago. We also thank B. Alvarez and E. H. Bucher for their valuable suggestions to improve the manuscript. This work was funded by Programa de Maestría en Manejo de Vida Silvestre (Universidad Nacional de Córdoba). We thank J. Brasca for improving the English style and G. Scrocchi, P. Cascivio and J. Cespedez for their help with species identification.

References

- Abril A, Bucher EH (1999) The effects of overgrazing on soil microbial community and fertility in the Chaco dry savannas of Argentina. *Appl Soil Ecol* 12:159–167
- Achaval F, Olmos A (2003) Anfibios y Reptiles del Uruguay. Graphis, Montevideo
- Andersen AN, Cook GD, Corbett LK, Douglas MM, Eager RW, Russell-Smith J, Setterfield SA, Williams RJ, Woinarski JCZ (2005) Fire frequency and biodiversity conservation in Australian tropical savannas: implications from the Kapalga fire experiment. *Austral Ecol* 30:155–167
- Ash AN (1995) Effects of clear-cutting on litter parameters in the southern Blue Ridge Mountains. *Castanea* 60:89–97
- Ash AN (1997) Disappearance and return of plethodontid salamanders to clearcut plots in the southern Blue Ridge Mountains. *Conserv Biol* 11:983–989
- Braunack MV, Walker J (1985) Recovery of some surface soil properties of ecological interest after sheep grazing in a semi-arid woodland. *Aust J Ecol* 10:451–460
- Brisson JA, Strasburg JL, Templeton AR (2003) Impact of fire management on the ecology of collared lizard (*Crotaphytus collaris*) populations living on the Ozark Plateau. *Anim Conserv* 6:247–254
- Bury RB (2004) Wildfire, fuel reduction, and herpetofaunas across diverse landscape mosaics in northwestern forests. *Conserv Biol* 18:968–975
- Cabrera A (1976) Regiones Fitogeográficas argentinas. Enciclopedia Argentina de Agricultura y Jardinería. ACME, Buenos Aires
- Canevari P, Blanc DE, Bucher EH, Castro G, Davison I (1998) Los Humedales de la Argentina. Clasificación, situación actual, conservación y legislación. Publicación N° 46. Wetlands International, Buenos Aires
- Cei JM (1980) Amphibians of Argentina. *Monitore Zoologico Italiano. Monografia* 2:1–609
- Cei JM (1993) Reptiles del noroeste, nordeste y este de Argentina. Herpetofauna de las Selvas Subtropicales, Puna y Pampas. *Mus Reg Sc Nat Torino Monograph* 14:1–949
- Colli GR, Bastos RP, Araujo AFB (2002) The character and dynamics of the Cerrado herpetofauna. In: Oliveira PS, Marquis RJ (eds) *The Cerrados of Brazil: ecology and natural history of a neotropical savanna*. Columbia University Press, New York, pp 223–241
- Cunningham S, Babb RD, Jones TR, Taubert BD, Veja R (2002) Reaction of lizard populations to a catastrophic wildfire in a central Arizona mountain range. *Biol Conserv* 107:192–201
- Duellman WE, Trueb L (1994) *Biology of amphibians*. The Johns Hopkins University Press, Baltimore
- Ensby M, Mathieson M, Smith GC (2000) A review of the effects of grazing and associated fire on biodiversity—an Australian perspective to inform and guide resource management and research in Queensland's forests. Technical paper 008, Department of Natural Resources, Indooroopilly, Queensland
- Fredericksen NJ, Fredericksen TS (2002) Terrestrial wildlife responses to logging and fire in a Bolivian tropical humid forest. *Biodivers Conserv* 11:27–38
- Gallardo JM (1987) *Anfibios Argentinos. Guía para su identificación*. Biblioteca Mosaico, Buenos Aires
- Gibson DJ, Hulbert LC (1987) Effects of fire, topography, and year to year climatic variation on species composition in tallgrass prairie. *Vegetation* 72:175–185
- Greenberg CH, Neary DG, Harris D (1994) A comparison of herpetofaunal sampling effectiveness of pitfall, single ended and double ended funnel traps used with drift fences. *J Herpetol* 28:319–324
- Heinrich ML, Kaufman DW (1985) Herpetofauna of the Konza Prairie Research Natural Area, Kansas. *Prairie Nature* 17:101–112
- James S, M'Closkey R (2003) Lizard microhabitat and fuel management. *Biol Conserv* 114:293–297
- Jones KB (1981) Effects of grazing on lizard abundance and diversity in western Arizona. *Southwest Nat* 26:107–115
- Jones KB (1988) Distribution and habitat associations of herpetofauna in Arizona: comparisons by habitat type. In: Szaro RC, Severson KE, Patton DR (eds) *Management of amphibians, reptiles, and small mammals in North America*, USDA Forest Service General Tech. Reports RM-166. Rocky Mountain Forest and Range Experiment Station, Fort Collins, pp 109–129
- Jones B, Fox SF, Leslie DM, Engle DM, Lochmiller RL (2000) Herpetofaunal responses to brush management with herbicide and fire. *J Range Manag* 53:154–158
- Jongman RHG, Braak CJ, Tongeren OFR (1995) *Data analysis in community and landscape ecology*. Cambridge University Press, Cambridge
- Kunst CR, Bravo S, Panigatti JL (2003) *Fuego en los ecosistemas argentinos*. Ediciones INTA. Santiago del Estero, Argentina
- Leynaud GC, Bucher EH (2005) Restoration of degraded Chaco woodlands: effects on reptile assemblages. *F Ecol Management* 213:384–390
- Magurran AE (1988) *Ecological diversity and its measurement*. Princeton University Press, Princeton
- Marone L (1990) Modifications of local and regional bird diversity after a fire in the Monte Desert, Argentina. *Rev Chil Hist Nat* 63:187–195
- Masters P (1996) The effects of fire-driven succession on reptiles in Spinifex grasslands at Uluru National Park, Northern Territory. *Wildl Res* 23:39–48
- Mc Cune B, Grace JB (2002) *Analysis of ecological communities*. Mj Software Design, Oregon
- Mc Garigal K, Cushman S, Stafford S (2000) *Multivariate statistics for wildlife and ecology research*. Springer, New York
- Mc Leod RE, Gates JE (1998) Response of herpetofaunal communities to forest cutting and burning at Chesapeake Farms, Maryland. *Am Midl Nat* 139:164–177
- Means DB, Campbell HW (1981) Effects of prescribed fire on amphibians and reptiles. In: Wood W (ed) *Prescribed fire and wildlife in southern forests*. Belle W. Baruch Forest Science Institute, Georgetown, pp 89–96
- Minshall GW, Robinson CT, Lawrence DE (1997) Postfire responses of lotic ecosystems in Yellowstone National Park, USA. *Can J Fish Aquat Sci* 54:2509–2525
- Moseley KR, Castleberry SB, Schweitzer SH (2003) Effects of prescribed fire on herpetofauna in bottomland hardwood forests. *Southeastern Naturalist* 4:475–486

- Mushinsky HR (1985) Fire and the Florida sandhill herpetofaunal community: with special attention to responses of *Cnemidophorus sexlineatus*. *Herpetologica* 41:333–342
- Mushinsky HR, Gibson DJ (1991) The influence of fire periodicity on habitat structure. In: Bell SS, Mc Coy ED, Mushinsky HR (eds) *Habitat structure: the physical arrangement of objects in space*. Chapman and Hall, London, pp 237–259
- Pelegri N, Chani JM, Echevarría AL, Bucher EH (2009) Effects of forest degradation on abundance and microhabitat selection by ground dwelling Chaco lizards. *Amphibia–Reptilia* 30:261–275
- Penn AM, Sherwin WB, Lunney D, Banks P (2003) The effects of a low-intensity fire on small mammals and lizards in a logged burn forest. *Wildl Research* 30:477–486
- Pianka ER (1996) Long term changes in lizard assemblages in the Great Victoria desert: dynamic habitat mosaics in response to wildfires. In: Cody ML, Smallwood JA (eds) *Long-term studies of vertebrate communities*. Academic, New York, pp 191–215
- Radke NJ, Wester DB, Perry G, Rideout-Hanzak S (2008) Short-term effects of prescribed fire on lizards in mesquite–Ashe juniper vegetation in central Texas. *J Appl Herpetol* 5:281–292
- Russell KR, Lear DHV, Guynn DC (1999) Prescribed fire effects on herpetofauna: review and management implications. *Wildl Soc Bull* 27:374–384
- Ryder RA (1980) Effects of grazing on bird habitats. In: De Graff RM, Tilghman NG (eds) *Management of western forest and grasslands for nongame birds*. U.S. Dept. of Agriculture Forest Service, Odgen, pp 61–66
- Saibene C, Montanelli S (1997) Mapeo de las comunidades vegetales leñosas del Parque Nacional Mburucuyá, Corrientes, Argentina. *FACENA* 13:49–57
- Schurbon JM, Fauth JE (2003) Effects of prescribed burning on amphibians diversity in a southeastern U S. national forest. *Conserv Biol* 17:1338–1349
- Scrocchi GJ, Giraudo AR (2005) Reptiles de la Reserva El Bagual. In: Di Giacomo AG, Krapovickas SF (eds) *Historia Natural y Paisaje de la Reserva El Bagual, Provincia de Formosa, Argentina: inventario de la fauna de vertebrados y de la flora vascular de un área protegida del Chaco Húmedo*. Aves Argentinas/Asociación Ornitológica del Plata, Buenos Aires, pp 155–198
- Sutherland WJ (1996) *Ecological census techniques: a handbook*. Cambridge University Press, Cambridge
- Vega LE, Bellagamba PJ, Fitzgerald LA (2000) Long-term effects of anthropogenic habitat disturbance on a lizard assemblage inhabiting coastal dunes in Argentina. *Can J Zool* 78:1653–1660
- Vitt LJ, Avila-Pires TC, Caldwell JP, Oliveira VR (1998) The impact of individual tree harvesting on thermal environments of lizards in Amazonian rain forest. *Conserv Biol* 12:654–664
- Vitt LJ, Colli G, Caldwell JP, Mesquita D, Garda AA, França FG (2007) Detecting variation in microhabitat use in low-diversity lizard assemblages across small-scale habitat gradients. *J Herpetol* 41:654–663
- Wester DB (1992) Viewpoint: replication, randomization, and statistics in range research. *J Range Manage* 45:285–290
- Whelan RJ (1995) *The ecology of fire*. Cambridge University Press, Cambridge
- Wilgers DJ, Horne EA (2006) Effects of different burn regimes on tallgrass prairie herpetofaunal species diversity and community composition in the Flint Hills, Kansas. *J Herpetol* 40:73–84
- Wright HA, Bailey AW (1982) *Fire ecology, United States and Southern Canada*. Wiley, New York
- Zar JH (1984) *Biostatistical analysis*. Prentice-Hall, New Jersey
- Zug GR, Vitt LJ, Caldwell JP (eds) (2001) *Herpetology: an introductory biology of amphibians and reptiles*. Academic, San Diego