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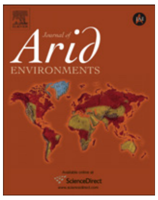

# Effects of habitat degradation on the lizard assemblage in the Arid Chaco, central Argentina

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## A B S T R A C T

Habitat degradation in the Chaco has been intense and widespread since European settlement. Detailed knowledge about its effects on the Chaco fauna is scarce despite the critical importance of such information for biodiversity conservation. The effects of habitat degradation on the Arid Chaco lizard assemblage were evaluated by comparing species richness and abundance in four sites along a gradient of increasing vegetation degradation. Two undisturbed sites (primary forest and grassland), and two sites with different degrees of forest degradation (moderate and severe) were selected. Ten lizard species were recorded (nine were trapped and one was sighted), totaling 1122 captures. Abundance of *Homonota fasciata*, *Leiosaurus paronae*, *Liolaemus chacoensis* and *Stenocercus doellojuradoi* was highest in the primary forest, with significantly lower values in the remaining three sites. *Teius teyou* and *Tropidurus etheridgei* were more abundant in the moderately and severely degraded forests than in the primary forest. Management implications of our results in terms of lizard biodiversity in the Chaco are analyzed.

## 1. Introduction

The Gran Chaco is a vast plain that extends from the tropics to the subtropics (between 18 °S and 31 °S), encompassing about 1.2 million km$^2$ in Paraguay, Bolivia, Argentina, and a small portion of Brazil. It covers about 1500 km from north to south, and about 700 km from east to west, without any important intervening physical barriers. The climate is semi-arid and monsoonal type, with rainfall concentrated in summer. Mean temperature rises from south to north (18 °C–26 °C) and annual rainfall, from west to east (400 mm–1200 mm) (Bucher, 1982). While the eastern portion of the Chaco (known as Eastern Chaco or Humid Chaco) is characterized by a savanna parkland and abundant wetlands, the Western Chaco shows a much drier landscape, dominated by a medium-tall xerophyllous, semi-deciduous subtropical woodland and presence of both edaphic and fire-maintained grasslands, particularly on sandy soils (Bucher, 1982). The western Chaco is currently divided into two subregions: the Semi-arid Chaco, which covers most of the region, and the Arid Chaco, restricted to the drier areas of the south-western border of the region (Secretaría de Desarrollo Sustentable y Política Ambiental and Ministerio de Desarrollo Social y Medio Ambiente, 1999).

Since European colonization, vegetation in the Semi-arid and Arid Chaco has changed drastically because of logging and overgrazing by domestic herbivores. Overgrazing led to fire suppression and a rapid and widespread shrub encroachment process. The final stage of the degradation process is a shrubland with sparse if any trees and almost absent herb and grass cover (Bucher, 1982; Bucher and Schofield, 1981; Kunst and Bravo, 2003; Morello and Saravia Toledo, 1959a).

Literature on degradation of Chaco vegetation is abundant (Bucher, 1982; Bucher and Schofield, 1981; Herrera et al., 2003; Morello and Saravia Toledo, 1959b); however, very little is known about the effects of habitat changes on the Chaco vertebrate fauna. This information is of critical importance for biodiversity conservation in terms of both the resulting changes in habitat suitability of species and disturbances in plant–animal interactions.

As in other semi-arid regions of the world, Chaco lizards are a conspicuous component of vertebrate biodiversity. Most studies on this animal group in the western Chaco have been conducted in the northern portion of the region (Semi-arid Chaco) (Cruz et al., 1992; Fitzgerald et al., 1999; Leynaud and Bucher, 2005; Pelegrin et al., 2009). Available literature on the Arid Chaco is scarce and

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refers to lizard distribution (Briguera et al., 2005; Pelegrin et al., 2006) and the effect of fire on lizard species (Pelegrin and Bucher, 2010).

We compared lizard species diversity and relative abundance in sites under different vegetation condition with the aim of evaluating the effects of Chaco land-use changes on lizard habitat suitability and the implications in terms of lizard biodiversity and abundance. Four specific vegetation conditions were selected along a gradient of increasing degradation: inside a protected area: a) primary forest, b) fire-generated grassland; outside a protected area: c) moderately degraded forest with a relatively well preserved tree canopy, and d) severely degraded forest.

## 2. Methods

### 2.1. Study area

The study was conducted in the southernmost portion of the Chaco (Arid Chaco subregion, Secretaría de Desarrollo Sustentable y Política Ambiental and Ministerio de Desarrollo Social y Medio Ambiente, 1999), west of Córdoba Province, central Argentina (Fig. 1). The study area includes Chancaní Provincial Natural Park and Natural Forest Reserve (hereafter Chancaní Reserve) (30°22′ S, 65°26′ W; 4960 ha) and neighboring areas. Chancaní Reserve is the only effectively protected fragment of the Arid Chaco forest in the entire region. Since its creation in 1986, the area was fenced and vegetation has remained free of human disturbance (Cabido and Pacha, 2002).

The local climate corresponds to the Cwa type under Köppen-Geiger classification system (Peel et al., 2007). Average annual rainfall is 425 mm, concentrated in the spring-summer months (November–March). Mean annual temperature is 18 °C, with a mean of 25 °C in January, the warmest month, and of 10 °C in July, the coldest month.

The dominant vegetation is dry semi-deciduous woodland. Canopy is discontinuous (15 m high), dominated by *Aspidosperma quebracho-blanco* and *Prosopis flexuosa*. The thorny, dense, and almost continuous shrub stratum (4 m high) is dominated by *Larrea divaricata*, *Mimozyganthus carinatus*, and *Acacia furcatispina*

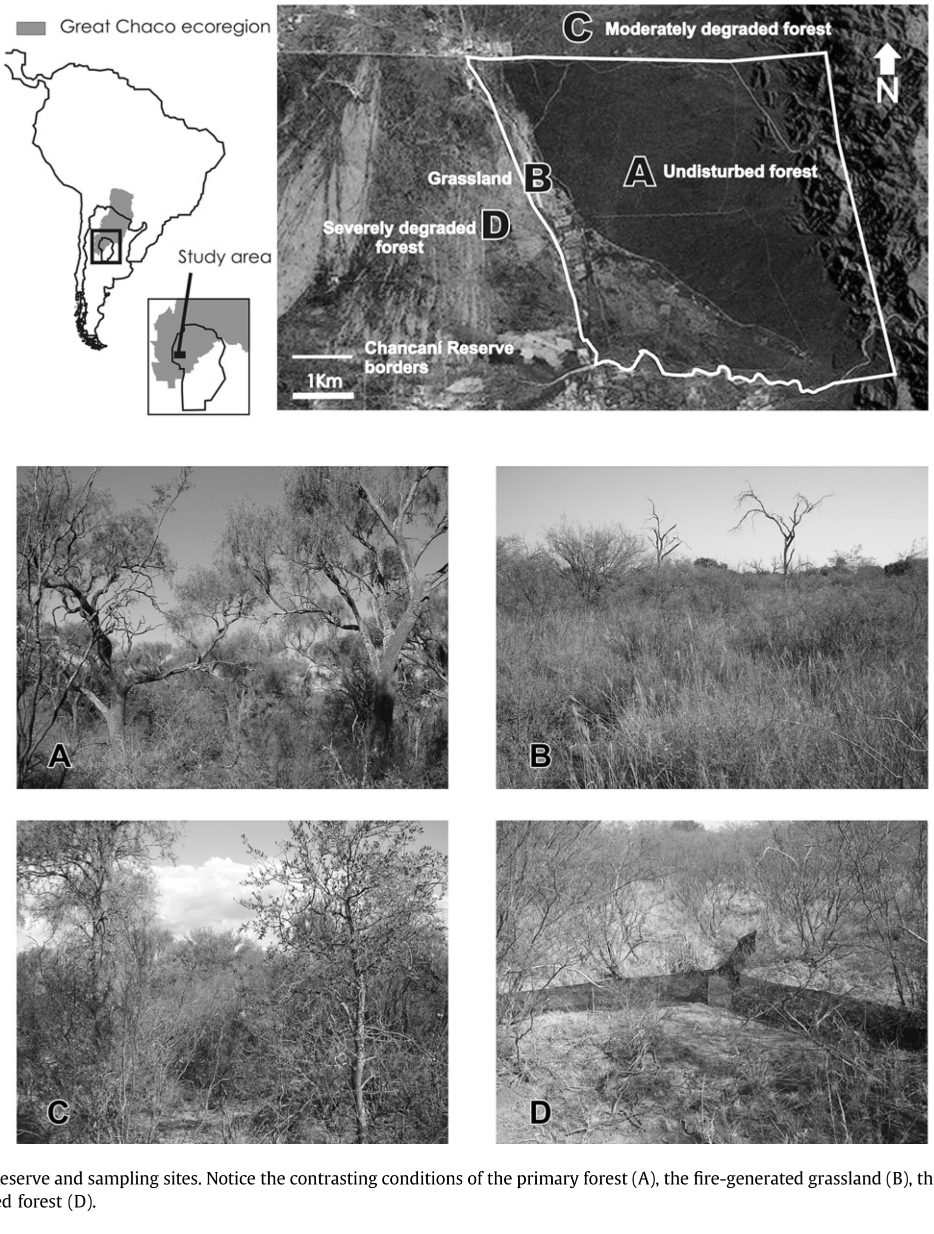

**Fig. 1.** Location of Chancaní Reserve and sampling sites. Notice the contrasting conditions of the primary forest (A), the fire-generated grassland (B), the moderately degraded forest (C), and the severely degraded forest (D).

(Cabido and Pacha, 2002). The lizard fauna in Chancaní Reserve includes 12 species (Pelegrin et al., 2006).

### 2.2. Experimental design

Four sites representing four vegetation conditions were selected (Fig. 1). Inside the Chancaní Reserve: 1) primary forest: a forest that has remained undisturbed for at least the last 50 years; 2) fire-generated grassland: a patch of grasses (>1.5 m tall) generated after a wildfire (1994), which remained undisturbed since then. Outside Chancaní Reserve: 3) moderately degraded forest: a forest with dominant, predominantly mature, tall trees under long-term cattle and goat grazing and selective logging; and 4) severely degraded forest: a highly degraded shrub-dominated area, with a long history of overgrazing by cattle and goat, also affected by the 1994 fire.

Because of the increasing degradation of the Arid Chaco, primary forest and grassland no longer exist outside the Chancaní Reserve. We assume therefore that at present vegetation within the Chancaní Reserve is unique in terms of species condition and vegetation structure, and that it can be considered very close to the pre-European Chaco forest condition, based on descriptions of the 18th century (Kurtz, 1887). Accordingly, no true replicates were available (Hurlbert, 1984). However, the following factors support the validity of our analysis: a) the sites have a known management history; b) they were up to 4 km apart from one another, thus removing the effect of climate variation, and c) they were sampled concurrently, canceling out temporal effects.

In each site, a plot 1000 m long and 300 m wide (3 ha) was established. The four plots established in the study area were at least 2 km apart. At each plot, 25 traps were assembled, totaling 100 traps across all four sites. Each trap consisted of one buried plastic bucket (18-L, 30 cm diameter, and 35 cm depth) from which three drift-fences departed in a Y-shaped arrangement (Fig. 1D). Each fence was 2 m long and 0.5 m high. Traps were distributed evenly in open-vegetated plots. In plots with closed, thorny vegetation, traps were located near internal trails opened up with a machete for that purpose or trails used by cattle. In all cases, a minimal distance of 30 m was kept between traps. Due to the lack of true replicates, traps were used as replicates for the purpose of this study.

Lizards were trapped during 86 non-consecutive days in the warm-rainy season (October–March) of two consecutive years (2006–2007 and 2007–2008), totaling 8600 trap-days. Traps were left inactive during the cold-dry season (April–September), based on a previous work that indicated no lizard activity in Chancaní during this period (Pelegrin, 2006). However, non-systematic searches were made during the non-sampling period to check for the presence of active lizards.

All traps in a plot (25) were checked on a single day, so it took four days to check all traps. Trapping effort was equal every month. Therefore, results are expressed as relative indication of population abundance (lizards.trap$^{-1}$). Trapped specimens were identified, measured (Snout-Vent Length, SVL), toe-clipped to avoid double counting, and later released. Individuals were assigned to the adult or sub-adult (juveniles and neonates) age classes, based on the shortest SVL recorded in sexually mature females, as reported in the available literature: *Homonota fasciata*: 39.1 mm (Cruz, 1994); *Liolaemus chacoensis*: 38.0 mm (Cruz and Ramirez Pinilla, 1996); *Teius teyou*: 76.1 mm (Cruz et al., 1999); and *Tropidurus etheridgei*: 49.9 mm (Cruz, 1997). For *Stenocercus doellojuradoi* and *Leiosaurus paronae* an arbitrary value was set as the minimum value for adults, based on the bimodal distribution of size frequencies (SVL, mm) of these species. The SVL category with the smallest value between modal classes (*S. doellojuradoi*: 45 mm, *L. paronae*: 64 mm) was considered the minimum SVL for adults.

### 2.3. Microhabitat characteristics

The following microhabitat characteristics were assessed at each trap location in the four plots: a) ground cover (percent cover of bare ground, leaf litter, and grass & herbs); b) density of fallen logs & branches; c) density of shrubs; and d) density of trees. The proportion of total area covered by bare ground, leaf litter, and grass & herbs was measured using four 1-m$^2$ frames, three at the end of each trap fence, and the remaining one near the central bucket. Cover measures were averaged to obtain a single value for each trap. Shrubs and fallen logs & branches were counted within a triangular 27-m$^2$ area whose vertices were the outermost points of each of the three trap fences. Density of trees was estimated by counting all trees with diameter at breast height >20 cm in a circular area of 10-m radius, centered in the bucket of the trap.

### 2.4. Statistical analysis

Shannon Wiener Diversity Index ($H$) and species richness (number of lizard species) were calculated for each trap and compared among plots using Kruskal Wallis test and Tukey post hoc test.

Before the analysis, the number of lizards captured at each trap throughout the study period was pooled to avoid problems associated with zero-inflated data. To compare relative abundance among species in each plot and of each species among plots, trap-specific counts of lizard captures obtained at each plot throughout the trapping period was the response variable used. Therefore, relative abundance of lizards was represented as number of lizard captures/trap (catch per unit effort -CPUE). Relative abundances of lizards were compared with ANOVA, and Holm-Sidak post hoc tests. When data do not adjusted to normal distribution, we used the Kruskal Wallis test and Tukey post hoc test.

For comparisons of microhabitat characteristics among plots based on trap values we performed an ANOVA, and Holm-Sidak post hoc tests. When data do not adjusted to normal distribution, we used the Kruskal Wallis test and Tukey post hoc test. All tests were performed using an alpha level of 0.05 with Sigmastat statistical software. Response variables used to compare microhabitat characteristics among plots were: percent cover of bare ground, leaf litter, and grass & herbs; number of shrubs/100 m$^2$; number of fallen logs & branches/100 m$^2$; and number of trees/100 m$^2$.

The relationship between lizard species and habitat characteristics was explored with a Canonical Correspondence Analysis (CCA). For CCA, the independent variables consisted of a matrix values for each habitat variable (bare soil, litter, grass & herb cover, density of logs & branches, and canopy cover) as columns, and trap number as rows. The dependent variables consisted of a matrix with abundances of lizard species (*S. doellojuradoi*, *T. teyou*, *L. chacoensis*, *H. fasciata*, *L. paronae*, and *T. etheridgei*) as columns and trap number as rows. We tested the null hypothesis of no relationship between habitat and species matrices using PCORD 4.5 (McCune and Mefford, 1999), with the following configuration: Hill's ordination method (Hill, 1979), scaling of scores optimized for species, trap scores set as linear combinations of habitat variables, Monte Carlo test with 1000 permutations.

## 3. Results

### 3.1. Lizard diversity and richness

We recorded a total of 11 species, nine of which were captured in traps – totaling 1122 lizards including 53 recaptures – and the remaining two (*Tropidurus spinulosus* and *Tupinambis rufescens*)

**Table 1**
List of lizard species and total species richness recorded at the sampled plots based on captures and field observations. CP: Captured, OB: Observed.

| Species | Primary forest | Fire-generated grassland | Moderately degraded forest | Severely degraded forest |
|---|---|---|---|---|
| *Homonota fasciata* | CP | CP | CP | CP |
| *Leiosaurus paronae* | CP, OB | CP | CP | CP |
| *Liolaemus aff. gracilis* | | | | CP |
| *Liolaemus chacoensis* | CP, OB | CP | CP, OB | CP, OB |
| *Mabuya dorsivittata* | | CP | | |
| *Stenocercus doellojuradoi* | CP, OB | CP | CP, OB | CP, OB |
| *Teius teyou* | CP, OB | CP | CP, OB | CP, OB |
| *Tropidurus etheridgei* | CP, OB | CP | CP, OB | CP, OB |
| *Vanzosaura rubricauda* | CP | CP | CP | CP |
| *Tropidurus spinulosus* | OB | | | |
| *Tupinambis rufescens* | OB | OB | OB | OB |
| Total species richness | 9 | 9 | 8 | 9 |

were sighted but not trapped (Table 1). Of the species recorded, nine were terrestrial, one (*T. etheridgei*) was terrestrial and arboreal, and only one (*T. spinulosus*) was exclusively arboreal. The total number of species recorded in each site ranged between eight and nine. Three species were recorded in a single plot and in low numbers: *T. spinulosus* in the primary forest plot, *Mabuya dorsivittata* in the fire-generated grassland plot, and *Liolaemus* sp. in the severely degraded forest plot.

Mean species richness was higher in the moderately degraded (mean species richness ± SE = 4.80 ± 0.16) and the primary forest plots (3.92 ± 0.24) than in the severely degraded forest (3.48 ± 0.26) and the fire-generated grassland plots (2.68 ± 0.26) (Kruskal–Wallis, $H = 31.1$, $P < 0.001$; Post hoc: Tukey, $P \leq 0.05$). Mean Shannon Wiener Diversity index was higher in both the primary ($H \pm \text{SE} = 1.15 \pm 0.07$) and the moderately degraded (1.33 ± 0.05) forest plots than in the severely degraded forest plot (0.98 ± 0.08) and in the fire-generated grassland plot (0.77 ± 0.09) (Kruskal–Wallis, $H = 24.295$, $P < 0.001$; Post hoc: Tukey, $P \leq 0.05$). The moderately degraded forest plot had the highest relative abundance value (lizards.trap$^{-1}$ ± SE = 18.76 ± 1.41). Relative abundance of the other plots was: primary forest plot = 12.08 ± 0.99, severely degraded forest plot = 9.08 ± 0.74, and the fire-generated grassland plot = 4.96 ± 0.54 (Kruskal–Wallis test, $H = 54.1$, $P < 0.001$; Post hoc: Tukey, $P \leq 0.05$). Relative abundance of adults between plots was higher in the primary (lizards.trap$^{-1}$ ± SE = 6.32 ± 0.67) and moderately degraded (7.88 ± 0.65) forest plots than in the fire-generated grassland (2.44 ± 0.41) and severely degraded forest (3.32 ± 0.36) plots (ANOVA, df = 3, $F = 22.323$, $P < 0.001$; Post hoc: Holm Sidak, $P \leq 0.05$).

Of the nine species captured, six showed significant differences in abundance among plots (Fig. 2). In the primary forest plot, *S. doellojuradoi*, *L. chacoensis*, and *T. teyou* were the most abundant species, with 80% of total captures (Kruskal–Wallis test, $H = 131.59$, df = 8, $P < 0.001$. Post hoc test: Tukey, $P \leq 0.05$). In the moderately degraded forest plot, *T. teyou*, *T. etheridgei* and *S. doellojuradoi* were the most abundant species (75% of total captures) (Kruskal–Wallis test, $H = 163.118$, df = 8, $P < 0.001$. Post hoc test: Tukey, $P \leq 0.05$). In the severely degraded forest plot, *T. teyou* was the dominant species (52% of total captures) (Kruskal–Wallis test, $H = 129.447$, df = 8, $P < 0.001$, Post hoc test: Tukey, $P \leq 0.05$). In the fire-generated grassland plot, *S. doellojuradoi* and *T. teyou* were most frequently captured species (Kruskal–Wallis test, $H = 101.444$, df = 8, $P < 0.001$, Post hoc test: Tukey, $P \leq 0.05$).

### 3.2. *Species-habitat associations*

Five of the six habitat variables measured were affected by habitat degradation (Table 2). Compared to the primary forest plot, litter cover, density of logs & branches, and density of trees were reduced in degraded plots, whereas grass & herb cover and density of shrubs were higher in degraded plots. The only difference between primary and moderately degraded forest plots was the higher cover of grass and herbs of the latter (Table 2).

The CCA performed on the basis of 1000 Monte Carlo permutations showed a significant correlation among species and habitat variables around each trap for the two canonical axes (CCA axis 1: $R = 0.545$, $P = 0.002$; CCA axis 2: $R = 0.578$, $P = 0.001$). Eigenvalues were significant for both axes (CCA axis 1: eigenvalue = 0.062, $P = 0.003$; CCA axis 2: eigenvalue = 0.050, $P = 0.001$).

CCA axis 1 was negatively correlated with litter cover ($R = -0.330$), grass & herb cover ($R = -0.421$), density of trees ($R = -0.306$), and density of fallen logs & branches ($R = -0.523$), and positively correlated with bare ground cover ($R = 0.837$). CCA axis 2 was positively correlated with grass & herb cover ($R = 0.871$) and density of shrubs ($R = 0.322$), and negatively correlated with litter cover ($R = -0.842$), density of fallen logs & branches ($R = -0.444$), density of trees ($R = -0.336$), and bare ground cover ($R = -0.200$). The following associations were observed: *H. fasciata* and *S. doellojuradoi* with litter cover and *L. paronae* with density of trees and density of shrubs (Fig. 2).

## 4. Discussion

Our results indicate that habitat changes induced by overgrazing and deforestation (at least within the intensity range observed in Chancaní) would not lead to local species extinction of any of the lizards species considered here, which are dominant components of the Arid Chaco assemblage. Species composition was similar across plots, with minor differences attributable to rare species that were not detected in all plots. However, our study also shows that changes in habitat characteristics induced significant variations in lizard relative abundance, depending on taxonomic groups. Species diversity and richness, as well as relative abundance, were higher in the forest plots (primary and moderately degraded forests) than in the non-forest plots (fire-generated grassland and severely degraded forest).

The presence of open patches intermingled within the forest appears to be a key factor in controlling coexistence of lizard species in the Chaco, given that this is one of the main axis explaining habitat segregation in terrestrial lizards. The influence of horizontal heterogeneity in vegetation structure on Chaco lizards is also supported by the existence of two well-defined adaptive groups: those species whose relative abundance peaks in the primary forest plot; and those species that show lower values in the primary forest than in the remaining habitat plots. The former group includes species adapted to closed vegetation and shady environments (*H. fasciata*, *L. paronae*, *L. chacoensis* and

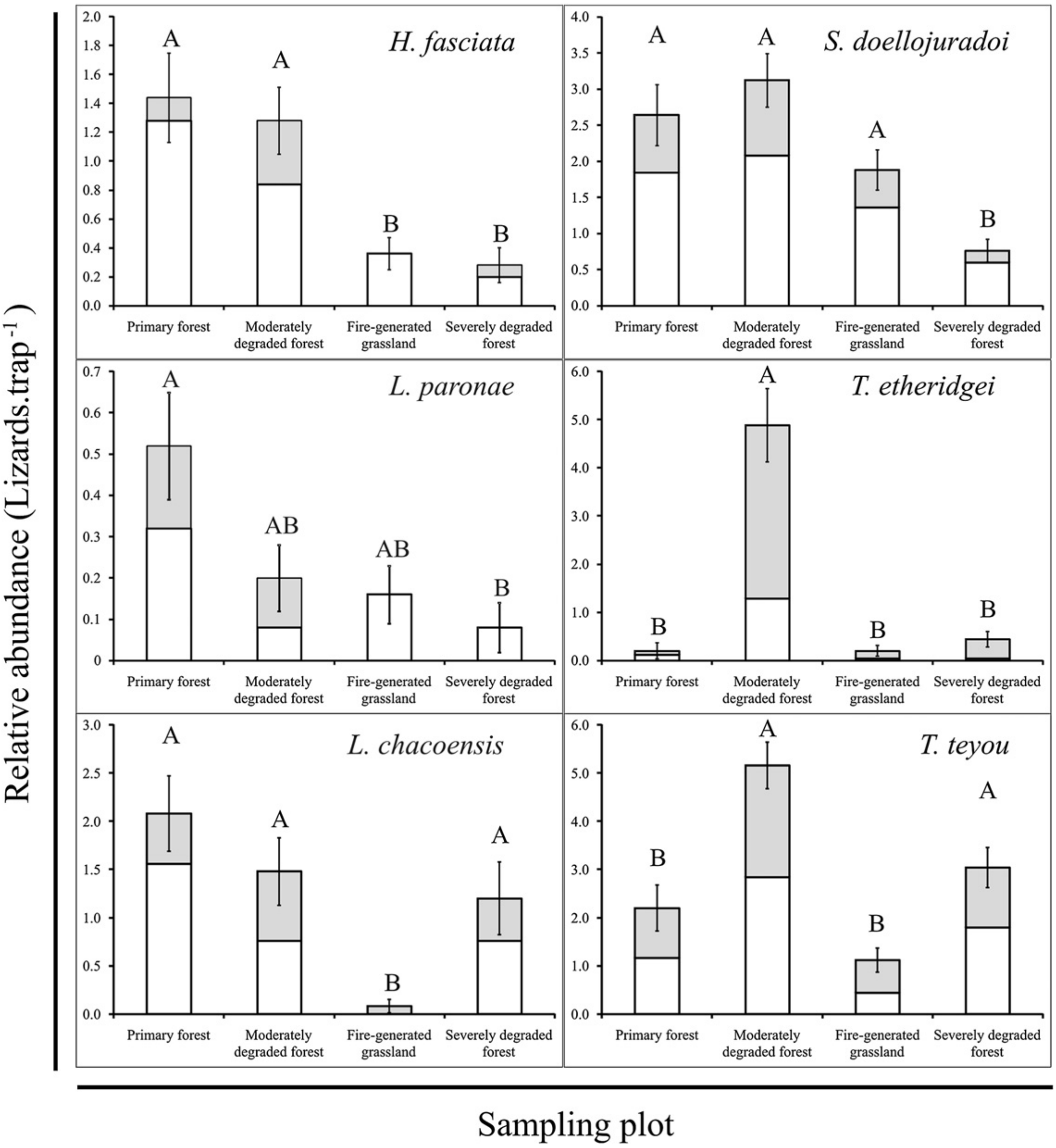

**Fig. 2.** Relative abundance of most abundant lizard species captured in the sampled plots. Different letters indicate significant differences in total (adults + juveniles) relative abundance among plots (Kruskal–Wallis test, $P \leq 0.05$, Post hoc: Tukey, $P \leq 0.05$). Error bars are shown. White bars: adults, gray bars: juveniles.

*S. doellojuradoi*), whereas the later includes typical heliothermic species that benefit from higher availability of open areas under direct solar radiation (*T. teyou*, *T. etheridgei* and *C. serranus*).

The influence of vegetation cover type on lizard assemblages observed in Chancaní is consistent with previous reports, indicating that high abundance of teiid lizards in open habitats is associated with low abundance of non-teiid and non-heliothermic lizard species (Sartorius et al., 1999; Vitt et al., 1998). It has also been reported that the occupation of forest openings by teiid lizards decreases with vegetation restoration, leading to a relaxation of

**Table 2**
Habitat characteristics of the sampled plots. Different letters indicate significant differences among plots (Kruskal–Wallis test, $P \leq 0.05$, Post hoc: Tukey test). Sampling size was 25 traps per plot.

| Habitat variable | Primary forest | Fire-generated grassland | Moderately degraded forest | Severely degraded forest | *H* | *P* |
|---|---|---|---|---|---|---|
| | Mean ± SE | Mean ± SE | Mean ± SE | Mean ± SE | | |
| Percentage cover (%) | | | | | | |
| Bare ground cover | 31.80 ± 2.42 | 29.89 ± 2.80 | 27.82 ± 2.89 | 35.83 ± 2.58 | 5.39 | 0.15 |
| Litter cover | 24.65 ± 2.51 A | 4.90 ± 1.25 B | 13.35 ± 2.18 A | 3.06 ± 0.76 B | 47.73 | <0.001 |
| Grass & herb cover | 9.15 ± 1.82 C | 47.23 ± 3.47 A | 25.76 ± 2.46 B | 52.60 ± 2.64 A | 64.83 | <0.001 |
| Trees (number. 100 $m^2$) | 2.96 ± 0.23 A | 0.54 ± 0.11 B | 3.30 ± 0.24 A | 1.29 ± 0.27 B | 52.67 | <0.001 |
| Shrubs (number. 100 $m^2$) | 12.141 ± 1.29 AB | 17.03 ± 1.91 AB | 11.85 ± 1.13 B | 18.07 ± 1.79 A | 10.46 | 0.02 |
| Fallen logs & branches (number. 100 $m^2$) | 37.48 ± 2.59 A | 21.92 ± 2.24 B | 43.55 ± 3.97 A | 10.37 ± 1.52 C | 55.31 | <0.001 |

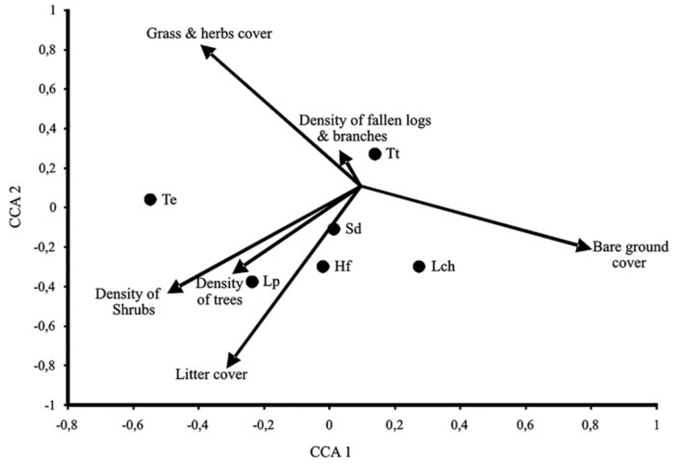

**Fig. 3.** Plot of Canonical Correspondence Analysis comparing matrices of microhabitat characteristics with lizard sampling data. Length of vectors indicates significance strength. Distance between points (species) and vectors (microhabitats) in the plot indicates the strength of the microhabitat-species association. References: Hf = *Homonota fasciata*, Lp = *Leiosaurus paronae*, Lch = *Liolaemus chacoensis*, Sd = *Stenocercus doellojuradoi*, Tt = *Teius teyou*, Te = *Tropidurus etheridgei*.

displacement of non-heliothermic lizards with time (Lima et al., 2001).

A special case is the fire-generated grassland, a much less favorable environment for lizards, which was occupied in low numbers by a few species associated with primary forest conditions. The homogeneous and dense grass cover may be a major limiting factor for lizards because a) tall grasses block direct solar radiation and may result in thermoregulatory constraints, and b) dense vegetation would make foraging difficult. A single species, *M. dorsivittata*, was found exclusively in this site, in very low numbers. It is abundant in tall grasslands in the Humid Chaco, under a much higher rainfall regime (Cano and Leynaud, 2010).

In terms of responses of the lizard assemblage to habitat changes, our results indicate several specific patterns. *L. paronae* showed a close association with the primary forest plot, where it reached highest relative abundance levels. This preference was confirmed by its association with density of trees and density of shrubs (Fig. 3), which may be related to the fact that *L. paronae* is phylogenetically related to arboreal Iguania (Cei, 1993). A similar pattern was reported for the Monte Desert (Mendoza, southwestern Argentina), where *L. paronae* was associated with tall shrubs and *Prosopis* trees (Videla and Puig, 1994).

*L. chacoensis* relative abundance also peaked in the primary forest plot, occurring at much lower level in the remaining woody vegetation habitats (moderately and severely degraded forest plots). *L. chacoensis* is the only Chaco representative of a highly diverse, typically heliothermic genus widely distributed in arid and semiarid habitats of South America (Abdala, 2007; Marquet et al., 1989; Schulte et al., 2003). In the Chaco it is associated with bare ground in primary and disturbed forests (Pelegrin et al., 2009), which may explain its presence in the severely degraded forest plot and its almost complete absence in the fire-generated grassland plot.

*H. fasciata* and *S. doellojuradoi* relative abundance was highest in forest-dominated habitats, both inside and outside the protected area. *H. fasciata* is a nocturnal gekkonid associated with litter cover, indicative of high tree cover under undisturbed conditions (Fig. 3). *S. doellojuradoi*, a little known Chaco endemic species, also shows association with litter cover. Its association with forests in good condition has been reported for the northern Chaco (Salta province) (Leynaud and Bucher, 2005; Pelegrin and Bucher, 2010). *S. doellojuradoi* also occurred at a high relative abundance in the fire-generated grassland plot, suggesting preference for microhabitats under dense woody or grassy vegetation cover.

The remaining two species (*T. etheridgei* and *T. teyou*) had peaks of relative abundance outside the primary forest of the protected area. *T. etheridgei* appeared with low relative abundance in all habitat plots except in the moderately degraded forest plot, where a marked peak was recorded. The latter record was primarily due to an exceptionally high abundance of juveniles at the moment of trapping in March. Although recruitment of juveniles occurred across the four plots, this exceptional peak of juvenile captures was observed only in the moderately degraded forest plot. When only adult population abundance was considered, no significant differences among plots became apparent (Fig. 2).

*T. teyou* was very abundant in all the studied habitats. Its preference for a wide range of habitats is supported by the lack of significant habitat associations shown in the CCA (Fig. 3). However, *T. teyou* seems to be favored by open areas, according to its higher abundance in forested plots with exploited woody vegetation (moderately degraded and severely degraded forests). This preference is also consistent with observations of *T. teyou* as a pioneer species in severely deforested Chaco areas in northern Argentina (J.M. Chani, pers. com.).

From the management point of view, lizard species assemblage does not appear as a highly reliable indicator of forest condition, because all the species studied occur under all the conditions evaluated, although in different abundances. However, some practical criteria may help improve monitoring tools of managed Chaco forest areas. Conditions close to the primary forest may be characterized by the presence of *H. fasciata*, *L. paronae*, *L. chaquensis*, and particularly *S. doellojuradoi*, in detectable numbers. By contrast, high relative abundance of *T. teyou* along with low levels of the previous species indicates a significant degree of forest disturbance. Similar responses have been documented in the Arid Chaco of Argentina (Salta and Córdoba provinces) (Leynaud and Bucher, 2005; Pelegrin and Bucher, 2010; Pelegrin et al., 2009).

Biodiversity conservation practices in Chancaní and other Arid Chaco protected areas should keep in mind, therefore, that the presence of lizard species that prefer disturbed habitats may be affected by management policies that aim at complete fire suppression, which leads to the elimination of fire-generated grasslands, a typical and significant Chaco habitat (Bucher, 1982).

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