Landsc Ecol (2025) 40:85
https://doi.org/10.1007/s10980-025-02096-y

RESEARCH ARTICLE

Agriculture cover and local vegetation structure shape Squamata's diversity in agricultural landscapes in Brazilian Cerrado

Gabryella de Sousa Mesquita · Priscila Silveira · Werther Pereira Ramalho · Juliana Silveira dos Santos · Iberê Farina Machado · Natan Medeiros Maciel · Wilian Vaz-Silva · Milton Cezar Ribeiro · Rosane Garcia Collevatti

Received: 1 December 2023 / Accepted: 4 April 2025 / Published online: 18 April 2025
© The Author(s) 2025

Abstract

Context Changes in landscapes due to the conversion of natural vegetation into agroecosystems are causing an alarming and rapid loss of biodiversity on a global scale.

Objectives We assess how local environmental factors and landscape structure influence Squamata reptiles' abundance, and alpha and beta diversities in agricultural landscapes in the Brazilian Cerrado.

Methods We sampled 20 sites using active visual search and pitfall traps to record Squamata reptiles' abundance and richness. For each sampling site, we measured seven variables of vegetation structure at local scale and calculated four landscape metrics at five spatial scales.

Results We recorded 145 individuals of 30 Squamata species comprising 15 lizards, 12 snakes, and three amphisbaenians. Agriculture cover had a negative effect on lizards' abundance and richness. No effect was found on the abundance and richness of snakes. Species turnover was the major process responsible for Squamata reptile dissimilarity among sites. Reptile total beta diversity and species turnover were influenced by variations in the number of trees at the local scale.

Gabryella de Sousa Mesquita and Priscila Silveira share the first authorship due to equal contribution.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s10980-025-02096-y.

G. d. Mesquita (✉) · N. M. Maciel
Laboratório de Herpetologia e Comportamento Animal, Programa de Pós-Graduação em Biodiversidade Animal, Universidade Federal de Goiás, Goiânia, Goiás, Brazil
e-mail: gabryella.mesquita@gmail.com

G. d. Mesquita · W. P. Ramalho · I. F. Machado
Instituto Boitatá de Etnobiologia e Conservação da Fauna, Goiânia, Goiás, Brazil

P. Silveira
Department of Animal Ecology and Systematics, IFZ, Justus-Liebig Universität, Giessen, Germany

P. Silveira · R. G. Collevatti
Laboratório de Genética & Biodiversidade, Universidade Federal de Goiás, Goiânia, Goiás, Brazil

W. P. Ramalho
Laboratório de Ecologia, Evolução e Sistemática de Vertebrados, Instituto Federal Goiano, Campus Rio Verde, Rio Verde, Goiás, Brazil

J. S. dos Santos · M. C. Ribeiro
Laboratório de Ecologia Espacial e Conservação, Departamento de Biodiversidade, Universidade Estadual Paulista, Rio Claro, São Paulo, Brazil

J. S. dos Santos
Museu de Biologia Prof. Mello Leitão, Instituto Nacional da Mata Atlântica, Santa Teresa, Brazil

W. Vaz-Silva
Escola de Ciências Médicas e da Vida, Pontifícia Universidade Católica de Goiás, Goiânia, Goiás, Brazil

Conclusions Our results highlight the negative effects of agriculture expansion on species abundance and richness in lizard communities. Moreover, vegetation structure may act as an environmental filter and predict the variation in Squamata reptiles' species turnover between sites. Our findings show that maintaining well-preserved patches of natural vegetation in agricultural landscapes is of utmost importance for Squamata reptiles' conservation and should be considered in landscape conservation and management design.

Keywords Alpha diversity · Beta diversity · Community structure · Lizard · Snake · Cerrado

Introduction

Agricultural landscapes now occupy ~ 40% of the world's land surface; therefore, understanding how such transformation affects biodiversity is one of the biggest challenges for conservation (Foley et al. 2005; Newbold et al. 2016). The unsustainable expansion and intensification of agriculture homogenize landscape structure (Kremen et al. 2012; Newbold et al. 2015), causing habitat loss and fragmentation and decrease connectivity between habitats (Fischer and Lindenmayer 2007). Habitat loss due to natural landscapes conversion into agricultural and human-made structures has led to a global and rapid biodiversity loss, especially in the megadiverse Neotropics (Rausch et al. 2019; Püttker et al. 2020). These changes cause biodiversity and ecological processes loss, directly affecting ecosystem function and services (Haddad et al. 2015; Newbold et al. 2015; Banks-Leite et al. 2020).

Habitat loss and fragmentation are intricately linked to changes in various landscape compositional (the type and abundance of land cover in space) and configurational (how land cover types are spatially distributed) characteristics that translate into a reduced amount and quality of habitat available for species and hinder species movements across the landscape, which can lead to ecological imbalances in ecosystems (With 2019). Edge effects (conditions that occur at the boundary of two distinct habitat types) also negatively impact vertebrate species' richness and abundance (Püttker et al. 2020). Therefore, these processes can alter species richness and composition of biological communities (Beca et al. 2017; Püttker et al. 2020; Cox et al. 2022; Luedtke et al. 2023), potentially triggering or accelerating extinction processes.

Terrestrial vertebrates are globally threatened by the sixth mass extinction crisis, mainly due to unsustainable agricultural practices (Munstermann et al. 2022), which often involve extensive replacement and split of natural soil cover, an essential part of the habitat of these organisms. For example, the decrease in landscape spatial heterogeneity (i.e., diversity of land use classes) and changes in habitat configuration (i.e., shape, size, and arrangement of habitat patches) in monoculture agroecosystems reduce species diversity in vertebrate communities such as in mammals (Regolin et al. 2020), birds (Dvořáková et al. 2023), and anurans (Silveira et al. 2023) worldwide. Given the intricate nature of landscape dynamics, it is crucial to deepen our understanding of how changes can affect vertebrate communities, particularly those historically overlooked, such as reptiles.

Reptiles are among the terrestrial vertebrates most threatened by human-induced landscape changes (Böhm et al. 2013; Keinath et al. 2017; Cox et al. 2022). The decline in reptiles increased sharply after the 1970s (Alroy 2015) due to habitat loss derived from agricultural expansion (Böhm et al. 2013). However, they are the most underrepresented vertebrates in studies analyzing the effects of landscape changes on community diversity, especially in non-forest ecoregions such as the Cerrado (Doherty et al. 2020; Teixido et al. 2020, 2021; López-Bedoya et al. 2022; Tan et al. 2023). The Brazilian Cerrado ecoregion is one of the world's hotspots of biodiversity (Myers et al. 2000), with more than 290 species of Squamata reptiles (Nogueira et al. 2011, 2019; Azevedo et al. 2016). However, since the 1980s, Cerrado has lost half of its native vegetation cover due to agricultural expansion (Alencar et al. 2020; Mapbiomas 2022), the most significant land-use change among Brazilian ecoregions (Strassburg et al. 2017).

In Cerrado, environmental filters and competition both at local (e.g., local vegetation structure) and landscape scales (e.g., percentage and fragmentation of natural vegetation in the landscape) are important determinants of herpetofauna community structure (Nogueira et al. 2005; Braga and Diniz 2015;

Ramalho et al. 2021; Colli et al. 2022). The causes of non-random community organization are varied and multi-scaled across time, space, and taxa (Arita and Rodríguez 2002; Viana et al. 2016), even for these ectothermic organisms (Buckley and Jetz 2008; Kay et al. 2018; Ramalho et al. 2021) whose communities were once thought to be randomly structured (Gotelli and McCabe 2002). Competition (Winemiller and Pianka 1990) and environmental filtering (Palmeirim et al. 2017b; González et al. 2022), including gradients imposed by land conversion to agriculture (Kay et al. 2018), are thus some of the claimed underlying causes of compositional differences in reptile communities. For example, agricultural intensification in India causes lower availability of forest patches negatively affecting herpetofauna diversity and abundance (Ghosh and Basu 2020). In Western Europe, agriculture intensification led to landscape simplification, which changed reptile distribution, restricting their occurrence to landscapes with dense hedgerows and forest edges, besides decreasing home ranges and population sizes (Guiller et al. 2022). In the Brazilian Amazon, forest structure and contrast in agricultural landscapes change reptile species composition and decrease forest specialist species abundance (Silva et al. 2022). In Cerrado, lizards are habitat specialists with little community overlapping between grasslands, savannas, and forests, resulting in habitat-structured lizard communities (Nogueira et al. 2009). Moreover, other studies have indicated variations in taxonomic, functional, and phylogenetic diversity, as well as in the assemblage structure, between naturally open and forest environments (Barros et al. 2022a, 2022b; Alexandre et al. 2023). Lizard communities in open formations tend to have higher taxonomic diversity but lower functional and phylogenetic diversity than those in forest formations (Barros et al. 2022a, 2022b). To our knowledge, no study analyzed the effects of agricultural landscapes and agriculture intensification in Cerrado's Squamata reptiles to date, which may compromise the conservation efforts aimed at protecting species within this taxon.

The environment is one of the main factors determining the diversity and structure of reptile communities (Moura et al. 2017). Therefore, local habitat and landscape structure play significant roles in generating dissimilarities among reptile communities from different habitats (Santos et al. 2016; Iglesias-Carrasco et al. 2022). Beta diversity, which quantifies the dissimilarity in species composition between communities (Whittaker 1972), is divided into two components: dissimilarity due to species replacement (species turnover) and dissimilarity arising from variations in species richness linked to the uneven distribution of resources, habitat patches discontinuities, or species loss (Baselga 2010; Moreno et al. 2018). Even though species turnover is expected to be the major driver of composition dissimilarities among reptile communities, variations in the prevalence of a specific pattern (turnover or nestedness) may rise due to a myriad of reasons (e.g., temporal or spatial variation of habitat heterogeneity, habitat connectivity, habitat disturbance, species traits and interactions, human-induced habitat modification). Therefore, this highlights the uniqueness of each ecosystem and represents an opportunity for ecologists to investigate further how environmental factors drive dissimilarities among biological communities, particularly in the context of rapid human-induced habitat modification. It is likely that in landscapes that have experienced the conversion of natural areas into agroecosystems, species spatial turnover can be the primary driver – but not exclusive – of dissimilarity among reptile communities at the local or landscape scale due to environmental filters derived from the extensive modification on the spatial distribution and amount of natural cover. Reptiles are highly sensitive to landscape changes due to their low dispersal capacity, morphological specialization in substrate type, and relatively small home ranges (Kearney et al. 2009; Böhm et al. 2013). Thus, habitat loss and changes in landscape composition can negatively affect the species richness, abundance, and composition of Squamata reptiles' communities by favoring generalist species in agricultural landscapes (Scott et al. 2006; D'Cruze and Kumar 2011). However, these topics have not been evaluated for this group in the Brazilian Cerrado in a multiscaled landscape context.

Here, we address the effects of landscape and local vegetation structure on the diversity of Squamata reptiles in the Brazilian Cerrado ecoregion. The replacement of natural soil cover by intensely managed agriculture fields and monoculture has been promoting dramatic structural and compositional homogeneization across the Cerrado, an ecoregion

originally covered by a highly heterogeneous vegetation. We study landscapes with intensive agricultural farming interspersed with natural vegetation remnants and a protected area (see details in Santos et al. 2021). Using a multiscale approach (Jackson and Fahrig 2015), we test the hypothesis that local vegetation and landscape structure influence Squamata communities' alpha and beta diversity. Reptiles exhibit diverse life-history strategies and habitat requirements, making them highly responsive to descriptors that structure the local vegetation, such as leaf litter biomass, canopy openness, and tree density. Therefore, we predict that local vegetation structure directly affects in reptile abundance and alpha diversity. At the landscape level, agricultural expansion leads to habitat loss, fragmentation, and landscape homogenization, which can negatively impact reptile diversity (Teixido et al. 2021; Sousa et al. 2022; Ramalho et al. 2022; Silveira et al. 2023). We, therefore, expect that landscapes with a higher amount of natural vegetation instead of agricultural cover, little fragmentation of natural vegetation (i.e., lower edge density), and higher compositional heterogeneity support greater reptile abundance and alpha diversity. Regarding species composition variation, we predict that species turnover is the primary component explaining the beta diversity between the study areas. Species turnover should be higher between sites that differ in local vegetation structure. At the landscape level, species replacement should be higher between landscapes with greater differences in the amount and fragmentation of natural vegetation and compositional heterogeneity.

Materials and methods

Study area and sampling design

We studied 20 agricultural landscapes in the LAND LTER (Agricultural landscape dynamics and impacts on biodiversity Long-Term Ecological Project program) in the Brazilian Cerrado (Fig. 1 and Table S1). The landscapes are dominated mainly by crops (mostly monocultures of soybean and corn) and pasturelands interspersed by remnants of natural vegetation, such as riparian forests, seasonally dry forests, savannas, forested savannas and wetlands (see details in Santos et al. 2021), and a protected area, Silvânia National Forest (FLONA) with 487.4 ha (Fig. 1).

We first created a land cover map at a 5 m spatial resolution using Google Earth images, freely available in the OpenLayer plugin of the Quantum Geographic Information System QGIS (QGis Development Team 2017). The classification was performed by visual interpretation and later validated in our study area. The resulting map included 13 land cover categories (Fig. 1). For landscape analyses, we merged riparian and seasonal forests and savannas into a single class to represent natural vegetation cover. Additionally, we merged the agriculture and pasture categories into one for the agricultural cover class. Next, we randomly selected 20 sampling sites to sample Squamata species and measure local vegetation structure and landscape variables. All sampling points were located within areas of native vegetation; four sites were within the protected area, and the remaining ones (n = 16) were in vegetation remnants of the Brazilian Cerrado in the study area.

Squamata community sampling

We sampled the Squamata community (lizards, snakes, and amphisbaenians) at 20 sampling sites over three field expeditions, lasting 10 days each, between September 2017 and May 2018. This resulted in a total of 30 sampling days. We conducted active searches limited by space and time (Martins and Oliveira 1998) at night, carrying out random walks throughout the natural vegetation fragments where we also installed pitfall traps (Cechin and Martins 2000; see Fig S1). While actively searching for individuals within a 100-m radius buffer departing from the centroid of each pitfall trap, we recorded encounters and collected all individuals to assess species richness and abundance. The sampling effort of this method was calculated by multiplying the number of sampling hours (18 h/site) by the number of researchers (three), totaling 1080 h. Each pitfall trap comprised five buckets of 60 L arranged in a straight line, interconnected through 0.4 × 5 m black plastic canvas. To calculate the sampling effort of this method, we multiplied the number of buckets (100) by the total sampling hours (240 h/expedition), totaling 72,000 h. We collected all specimens found except large snakes (e.g., *Boa constrictor*) or venomous ones (e.g., *Bothrops moojeni* and *Crotalus durissus*) to avoid snakebite incidents.

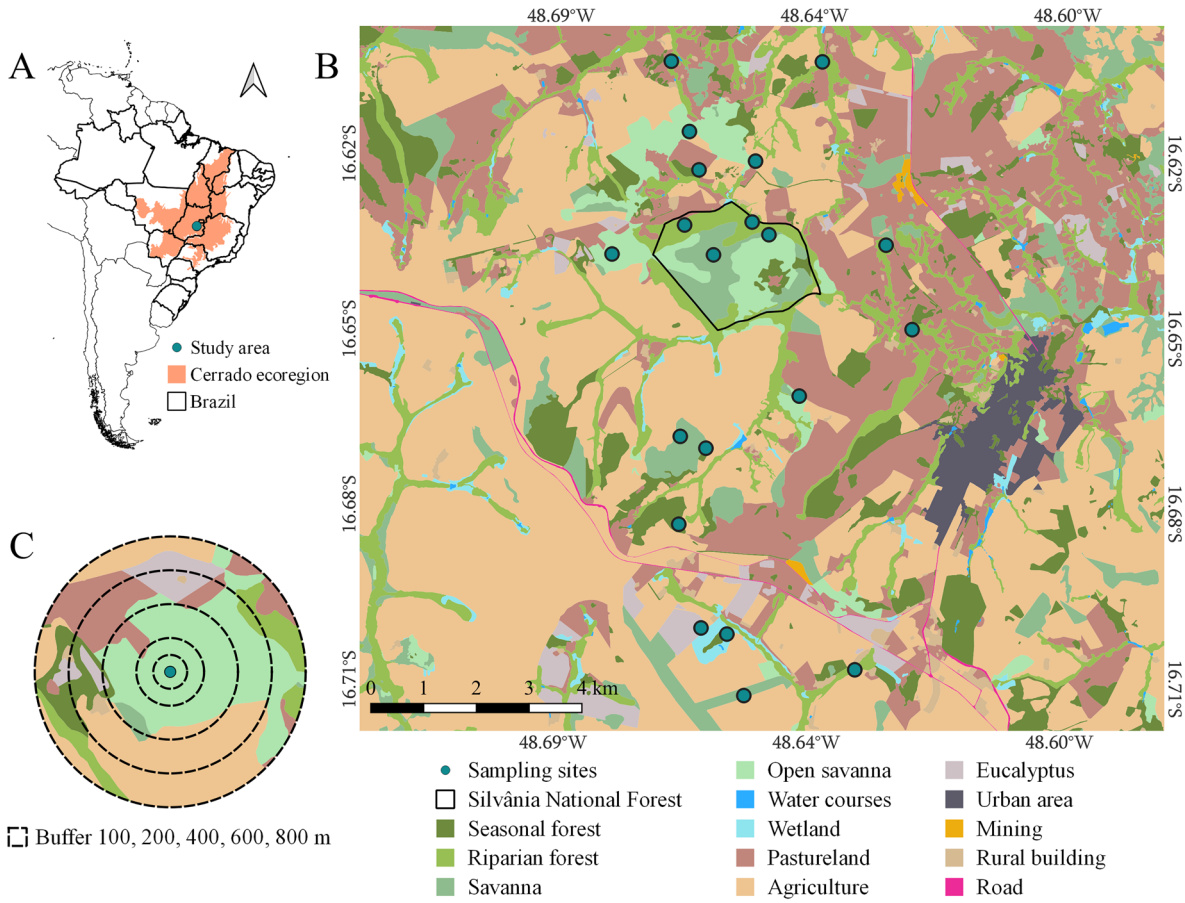


Fig. 1 Geographic distribution of the 20 sampling sites and land cover classes in the LAND LTER landscape. **A** The Brazilian Cerrado boundary (in orange) and the study area. **B** The LAND LTER study area with land cover composition and the 20 sampling sites. The protected area (Silvânia National Forest) is delimited on the map with a black continuous line. **C** Multiscale sampling design used to calculate the landscape metrics, represented by buffers around the landscape midpoint

Voucher specimens were obtained by first anesthetizing individuals, which were then killed with an injection of 5% xylocaine and fixed in 10% formalin before being preserved in 70% ethanol and deposited in the Coleção Herpetológica do Centro de Estudos e Pesquisas Biológicas (CEPB) of Pontifícia Universidade Católica de Goiás (PUC-GO), Goiânia, Goiás, Brazil. The taxonomy used in this study follows the List of Brazilian reptiles (Guedes et al. 2022).

Local and landscape scale variables

At local scale, we measured variables that represent the vegetation structure along a gradient of the study site that ranges from open to forested vegetation and potentially influence the distribution and composition of Squamata communities (see Nogueira et al. 2005; Kanowski et al. 2006; Garden et al. 2007). In each site, we randomly selected three 5×5 m plots 10 m away from the pitfall trap and 25 m from each other, totaling 60 plots (see Fig. S1; adapted from Pereira et al. 2016). We then calculated the amount of (1) herbs (HER), (2) shrubs (SHR), (3) trees (TRE), (4) grasses (GRA), (5) canopy openness (COP), (6) tree diameter at chest height (DBH), and (7) leaf litter biomass (LLB). We determined the number of herbs, shrubs, and trees by counting the number of individuals of each growth form in the plot. We assessed grass cover by calculating the percentage of grass coverage in the plot. We used a densitometer to estimate canopy openness in percentage, and DBH was measured using metric tape. Additionally, we collected

and dried three leaf litter samples per plot within a quadrant of 50 × 50 cm to obtain the dried leaf litter weight (Martins et al. 2021).

We calculated variables that represent landscape structure to evaluate the effects of habitat loss and fragmentation on reptile communities within spatial buffers of 100, 200, 400, 600, and 800 m radius size around each sampling site. We calculated the percentage of natural vegetation (forest and savanna; %NV) as a proxy of habitat amount, and percentage of agriculture in the landscape including pasture, soybean, and corn (%AGR). We calculated edge density (ED) as a proxy of landscape fragmentation, considering all patches of the natural vegetation class. Landscape compositional heterogeneity was calculated using Shannon's diversity index (SHDI; McGarigal et al. 2012). All landscape metrics were obtained using the package *landscapemetrics* (Hesselbarth et al. 2019) in R version 4.1.3 (R Core Team 2022).

Abundance, alpha and beta diversity

We first separated lizards and snakes into two different datasets to calculate our response variables. Amphisbaenians were included in the lizard dataset because they are sister taxa (Pyron et al. 2013; Longrich et al. 2015; Pyron 2017). Moreover, amphisbaenian abundance and species richness were very low in our sampling sites (3 species and 5 individuals).

We calculated species abundance and observed species richness using the number of observed individuals because our sampling effort and sampling area size were the same across sampling sites. To account for undetected species, we used an extrapolation and interpolation technique to estimate species richness derived from Hill numbers of order 1 (q = 1) based on the species abundance using function iNEXT4steps available in the iNEXT4steps package (Chao and Hu 2024; Fig. S2). Hill numbers of order q = 1 are equivalent to Shannon diversity (Chao et al. 2014; Chao et al. 2020). Since the rarefaction curve did not reach a plateau and reptile data was sparse, we used the estimated species richness at a standardized sample coverage value (SC = 0.45) so that the estimated richness of each site could be compared across sites (Chao et al. 2020). In the iNEXT4steps function, the sample coverage is adjusted automatically to the minimum sample coverage shared among all samples when sample sizes are extrapolated to the double size (Chao et al. 2020). This procedure ensures comparability among communities and prevents high uncertainty in richness estimates (Chao et al. 2020).

For beta diversity, we calculated Sørensen pairwise dissimilarity (βsor) and partitioned it into the components species turnover (Simpson dissimilarity index, βsim) and nestedness (βnes) following Baselga (2010, 2013). Because of the abundance disparity among lizard and snakes, we applied the Hellinger standardization using the *decostand* function in the vegan package prior to calculating the beta diversity index, reducing the effect of large abundances (Oksanen et al. 2020). The averages of βsor, βsim, and βnes were obtained from the triangular matrices and graphically represented using ternary plots to visualize the trends of the turnover and nestedness components of the total beta diversity. Beta diversity indices were computed with the *beta.div.comp* function from *adespatial* package (Dray et al. 2023) and ternary plots with triangle.plot function from ade package (Thioulouse et al. 2018) in R version 4.1.3 (R Core team 2024). Sites without species records were excluded from the analysis since the R packages we used do not accept sites with no species records.

Statistical analysis

Scale of effect of landscape variables

To evaluate the effects of landscape structure on the abundance and alpha diversity of lizards and snakes, we first analyzed the scale of effect to find the spatial extent where the biological response variable is best predicted by landscape structure. This is a crucial step when conducting landscape analysis because the strength and even the direction of the relationship between response variables and landscape predictors may vary depending on the scale that predictors were measured (Holland 2004; Jackson and Fahrig 2015). To do so, we regressed each response variable against landscape predictors at each spatial scale separately using the *multifit* function implemented in R version 4.1.3 (Huais 2018; R Core team 2022) and selected the spatial scale of each landscape variable with the lowest AIC value (Tables S2 and S3). Note that the *multifit* function does not provide AICc values. The

selected spatial scales for each landscape predictor of species richness and abundance are shown in Table 1. We used the landscape predictors of beta-diversity at the spatial scales obtained for species richness.

Abundance and Alpha diversity

We transformed both local vegetation and landscape variables into Z-scores to standardize and normalize predictors (see Cohen et al. 2003). Then, for each response variable (abundance and species richness), we built two full models: a full model including all local predictors and a second one with all landscape variables to test statistical assumptions and multicollinearity. Subsequently, for each model we assessed multicollinearity among predictors using the variance inflation factor (VIF) with the *check_collinearity* function. We kept only those predictors with VIF < 5 (Zuur et al. 2010) in each model (Tables S4–S7). For each response variable (species abundance and observed richness), we used generalized linear models (GLM) with Poisson distribution and for estimated species richness we used glms with a Gaussian distribution. We checked whether model assumptions were met by first analyzing overdispersion applying the *check_overdispersion* function implemented in the R package *performance* (Lüdecke et al. 2021). When overdispersion was detected, we used GLM with negative binomial distribution. We also checked for residuals' autocorrelation using GLM with the *check_autocorrelation* function also available in the performance R package (Lüdecke et al. 2021). Model residuals were independent and not autocorrelated ($P > 0.05$). We then performed model selection to evaluate the importance of local vegetation structure and landscape variables (Table 1) on abundance and diversity of lizards and snakes.

Finally, to find which variables best explained variation in abundance and species richness between sites, we built univariate models using the remaining uncorrelated variables (see Tables S6 and S7 for all tested models). We investigate the relationship between squamata diversity and landscape predictors by building univariate models to avoid model overparameterization and loss of degrees of freedom given the number of sample sites ($n = 20$). We also built a null model only with the intercept to compete against the univariate models (model competition approach in Tables S8 and S9). We used Akaike's information criterion corrected for small sample sizes (AICc; Burnham and Anderson 2002) to identify the best-fitting models. AICcs values for each model were obtained with the function *ICtab* from the *bbmle* package (Bolker 2008). Models with $dAIC \leq 2$ were considered equally plausible to explain variation in the data (Zuur et al. 2010). We also calculated wAIC weight, which is the relative importance of each model to explain the observed variation in the response variables. We then assessed spatial autocorrelation for the residuals of the top-ranked models by calculating Moran's I for lag distances at a broader (regional) and a smaller (local) spatial scale. First, we calculated Moran's index for lag distances across the study area to include all sample sites (regional scale) using the *nb2listw* function from the *ncf* package (Bjornstad 2022). Secondly, we performed the same analysis encompassing sample sites within a geographic range that encompassed two-thirds of the total extension of the study area (local scale). We did not find significant spatial autocorrelation in the models' residuals (Figs. S3, S4 and S5; Table S10). The effect size of the selected models was accessed using the function *effectsize* from the *effectsize* package (Ben-Shachar et al. 2020).

Table 1 Selected scale of effect for each landscape variable used on models to explain reptiles' abundance and alpha diversity in the Brazilian Cerrado

Species group	Response variable	Landscape metrics			
		%NV	%AGR	ED	SHDI
Lizard	Abundance	100 m	400 m	800 m	800 m
	Richness	100 m	600 m	800 m	400 m
	Shannon diversity	400 m	800 m	800 m	400 m
Snake	Abundance	600 m	600 m	400 m	600 m
	Richness	800 m	200 m	100 m	600 m
	Shannon diversity	800 m	200 m	100 m	600 m

%NV percentage of natural vegetation, *%AGR* percentage of agriculture, *ED* edge density, *SHDI* landscape compositional heterogeneity

Beta diversity

We calculated the ratio between βnes and βsor (βratio) as an indicator of the relative importance of turnover and nestedness. Values of βratio < 0.5 indicated that beta diversity is determined mainly by species turnover, whereas values > 0.5 represents that nestedness is the most important component (Dobrovolski et al. 2012). To assess the significance of βratio, we estimated the distribution of βratio by taking 999 random samples from 18 to 15 sites for lizards and snake communities, respectively, using a null model that randomizes community data matrix maintaining species occurrence frequency and site richness (Gotelli 2000). Next, we assessed whether βratio was larger or smaller than 0.5, using a p-value corresponding to the probability of obtaining the opposite result by chance. We checked for spatial autocorrelation in reptiles' community composition (lizards and snakes, separately) with Moran's Eigenvector Maps (MEM) using the *listw.candidates* function to create a list of spatial weighting matrices, and *listw.select* to select the best subset of eigenvectors. Both functions are from the *adespatial* package (Dray et al. 2023). We found no significant spatial autocorrelation in community composition (P > 0.05); thus, we included no spatial predictors in our models.

We assessed the relationships between βsor, βsim, and βnes (Tables S11-S12) and the dissimilarity matrices of local vegetation structure and landscape variables between pairs of communities using a distance-based redundancy analysis (dbRDA, Legendre 2014) with the *capscale* function in the *vegan* package (Oksanen et al. 2020). Then, for each response variable (βsor, βsim, and βnes), we built two full models: a full model including all local predictors and a second one with all landscape variables to test statistical assumptions and multicollinearity. The full models included all local vegetation structure and landscape variables at the spatial scale found to be the scale of effect of the variable on reptiles' richness (Table 1). We checked for multicollinearity among dbRDA model predictors using the variance inflation factor (VIF) with the *vif.cca* function available in the *vegan* package (Oksanen et al. 2020). We analyzed local vegetation structure and landscape variables separately. We kept only those predictors with VIF < 5 (Zuur et al. 2010) for the final models (Tables S12, S13). We then performed forward stepwise selection based on the adjusted R^2 criterion for each beta diversity component as well as for total beta diversity using the *ordiR2step* function in the *vegan* package (Oksanen et al. 2020) to identify which model, including the null model, would better explained the total beta diversity, turnover, and nestedness components. With this procedure, variables are only added to the model if they maximize the adjusted R^2 at each step, resulting in an overall model. Finally, we tested the significance of each overall model, variables, and canonical axes (CAP) using analyses of variance with the function *anova* in *vegan* package (R Core Team 2022).

Results

We recorded 145 individuals from 30 Squamata species comprising 15 lizards, 12 snakes, and three amphisbaenians. Reptile abundance ranged from zero to 19 individuals and richness from zero to nine species (Table S1). The most abundant species were the lizards *Norops brasiliensis* (n = 27), *Tropidurus torquatus* (n = 17), *Cercosaura schreibersii* (n = 9), *Colobosaura modesta* (n = 9), and *Notomabuya frenata* (n = 9). The rarest species were the snakes *Amerotyphlops brongersmianus*, *Apostolepis assimilis*, *Boa constrictor*, *Bothrops marmoratus*, *Crotalus durissus*, *Dipsas indica*, and *Dipsas mikanii*, represented by only one individual each in our samples.

Influence of local vegetation and landscape structure on species abundance and alpha diversity

Agriculture cover (%AGR) at 400 m was the best predictor of lizard's abundance (wAICc = 0.885; effect size: − 0.61; Table S8) and richness (wAICc = 0.776; effect size: − 0.62; Table S8). Lizard abundance and richness decreased as the percentage of agricultural cover increased (Fig. 2). Canopy openness was the best predictor for Shannon diversity of lizards (wAICc = 0.914; effect size: 0.70; Table S8). As canopy openness increases, the lizards' Shannon diversity is greater (Fig. 3). The null model was equally plausible to explain snake abundance and alpha diversity variation, thus suggesting that the influence of

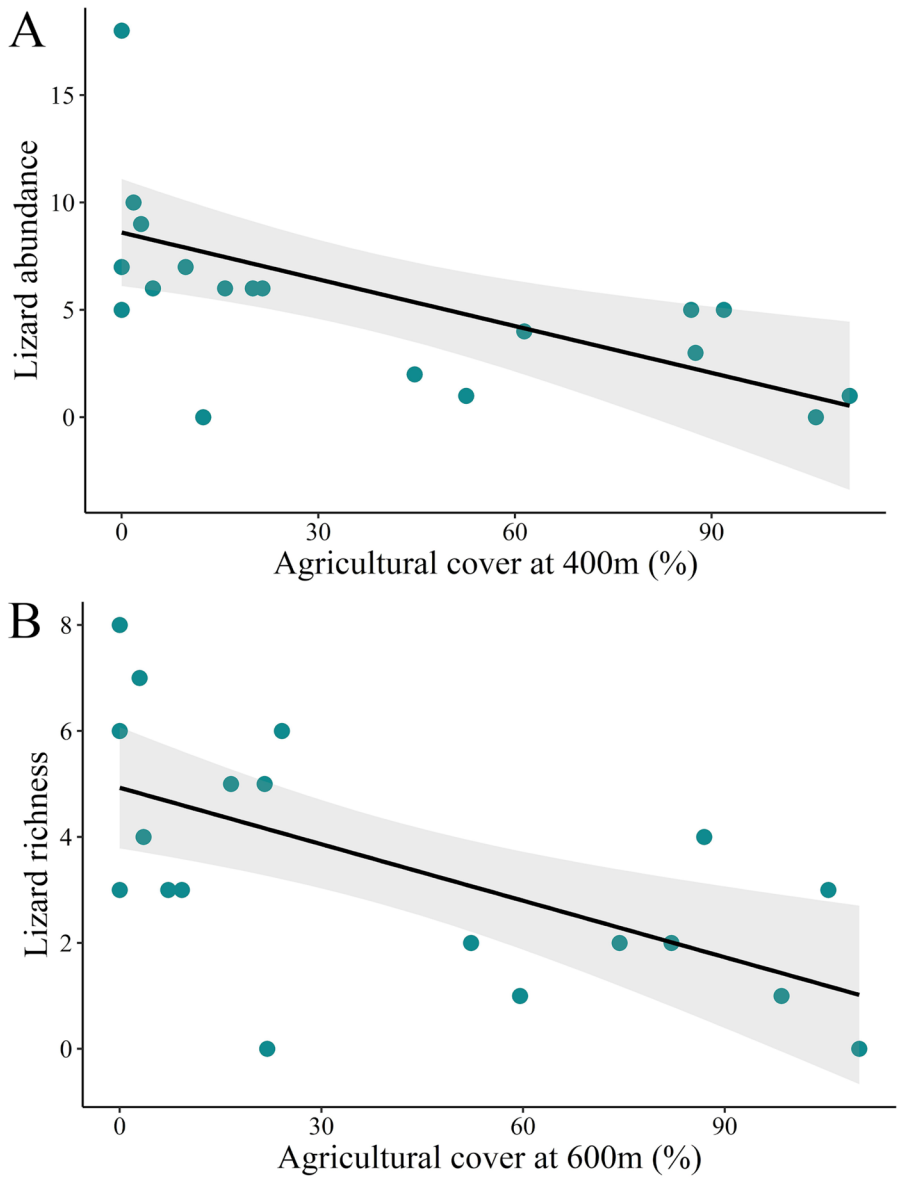


Fig. 2 Relationship between lizards' abundance and observed richness with agricultural cover (%) in 20 agricultural landscapes in the Brazilian Cerrado. **A** Relationship between species' abundance and % agricultural cover at 400 m spatial scale; **B** Relationship between species' richness and % agricultural cover at 600 m spatial scale. The solid line corresponds to the linear regression fit, and the shaded area represents the 95% confidence interval

local vegetation and landscape structure can also be attributed to chance (Table S9).

Influence of local vegetation and landscape structure on beta diversity

Squamata communities showed relatively high beta diversity (low similarity), sharing on average 24% and 10% of lizard ($\beta sor = 0.76$) and snake ($\beta sor = 0.90$) species, respectively. Species turnover was the predominant component, responsible for 94% of the variation in the species composition of lizards ($\beta sim = 0.71$) and 98% of snakes ($\beta sim = 0.88$) between sites (Fig. 4). $\beta ratio$ values reinforced this turnover pattern (lizards: $\beta ratio = 0.058$; snakes: $\beta ratio = 0.016$) and were significantly lower ($P < 0.005$) than expected by chance for both lizards ($\beta ratio = 0.154 \pm 0.023$) and snakes ($\beta ratio = 0.05 \pm 0.01$).

Db-RDA models showed that the number of trees ($F_{lizard} = 2.804$, $P_{lizard} = 0.005$; $F_{snake} = 2.103$, $P_{snake} = 0.021$; Figs. 5, 6, S6 and S7) was the main local variable explaining the total beta diversity of both lizard ($R^2adj = 0.113$, $F = 2.804$, $P = 0.009$) and

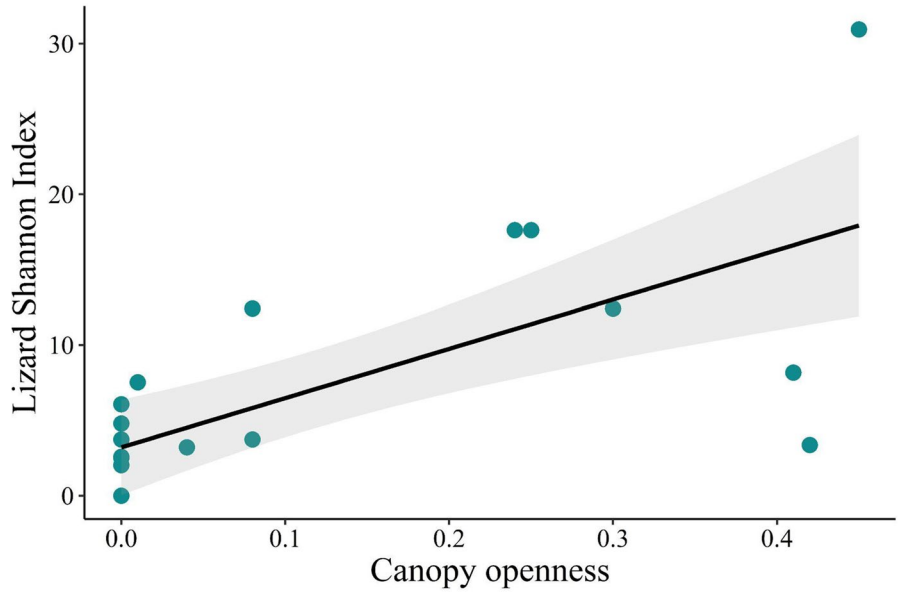


Fig. 3 Relationship between lizards diversity (q = 1) with canopy openness in 20 agricultural landscapes in the Brazilian Cerrado. The solid line corresponds to the linear regression fit, and the shaded area represents the 95% confidence interval

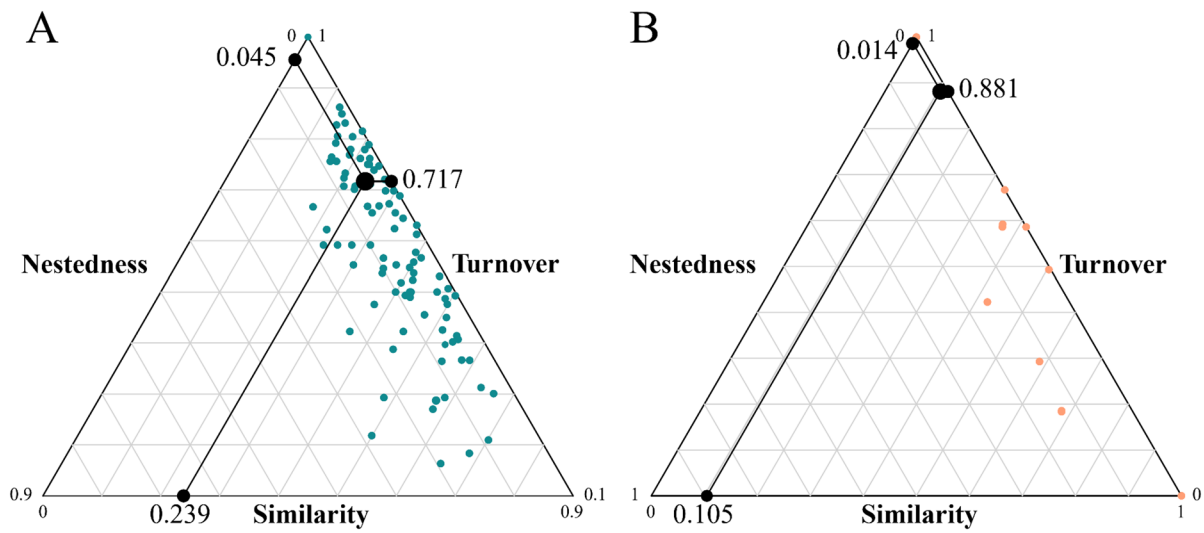


Fig. 4 Ternary plot of the relationship between pairs of sites sampled in the 20 agricultural landscapes in the Brazilian Cerrado for **A** lizards and **B** snakes. Each point represents a pair of sites, and their position is determined by the combination of similarity (1—dissimilarity), turnover, and nestedness, based on the Sørensen index of the Baselga's family

snake (R^2adj = 0.077, F = 2.103, P = 0.033) communities. Beta diversity was almost entirely explained by the turnover component (R^2adj_{lizard} = 0.132, F_{lizard} = 2.862, P_{lizard} = 0.013; R^2adj_{snake} = 0.077, F_{snake} = 2.151, P_{snake} = 0.027), which explained better the observed changes in community composition between sites for both lizard and snake, also attributed to the differences in tree amount (F_{lizard} = 2.862, P_{lizard} = 0.014; F_{snake} = 2.151, P_{snake} = 0.024).

Only the first axis of beta diversity db-RDA was statistically significant for both lizards (F = 2.804, P = 0.018) and snakes (F = 2.103, P = 0.022), explaining 15% and 14%, respectively, of the dissimilarity among communities. Likewise, only

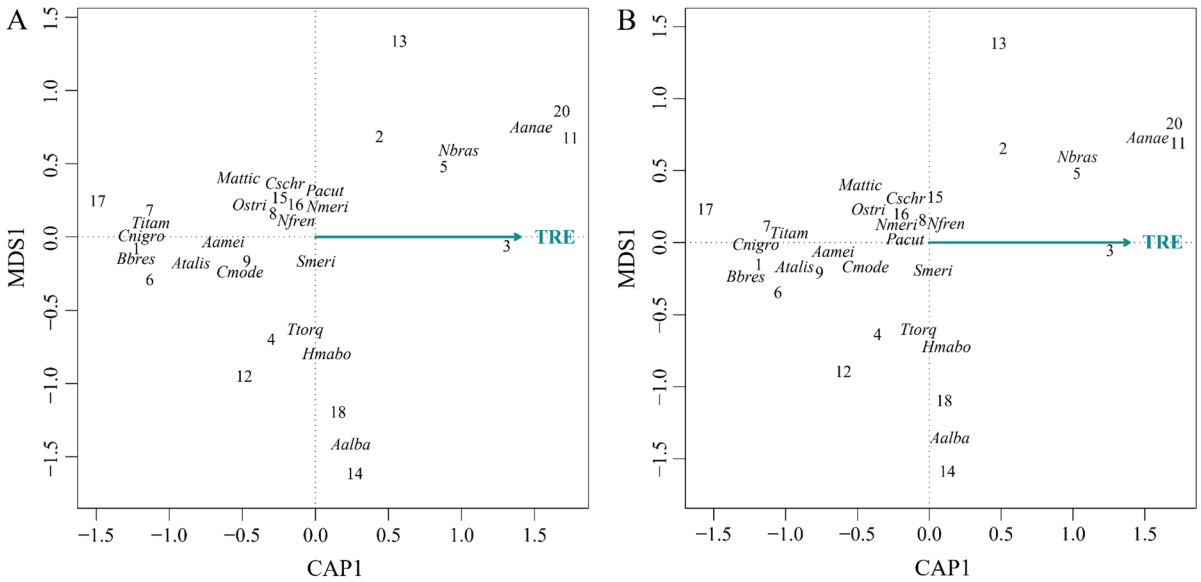


Fig. 5 Effect of number of trees (TRE) along the principal coordinate axis (CAP1) for the lizard's communities beta diversity (**A**) and its turnover component (**B**), based on redundancy analysis (db-RDA). MDS1 represents the second axis that was not significant in db-RDA analysis. Arrows indicate selected variables and the direction of effect. Aamei: *Ameiva ameiva*; Aalba: *Amphisbaena alba*; Aanae: *Amphisbaena anaemariae*; Atalis: *Amphisbaena talisiae*; Bbres: *Bachia bresslaui*; Cschr: *Cercosaura schreibersii*; Cmode: *Colobosaura modesta*; Cnigro: *Copeoglossum nigropunctatum*; Hmabo: *Hemidactylus mabouia*; Mattic: *Micrablepharus atticolus*; Nbras: *Norops brasiliensis*; Nmeri: *Norops meridionalis*; Nfren: *Notomabuya frenata*; Ostri: *Ophiodes striatus*; Pacut: *Polychrus acutirostris*; Smeri: *Salvator merianae*; Titam: *Tropidurus itambere*; Ttorq: *Tropidurus torquatus*

the first axis explained species turnover for lizards ($F=2.862$, $P=0.012$) and snakes ($F=2.151$, $P=0.033$), and beta diversity and turnover showed similar ordination patterns for both groups (Figs. 5 and 6). Thus, lizard and snake species are distributed across communities following the gradient created by differences in tree number.

Discussion

Here, we assessed how local vegetation and landscape structure affect reptile communities in agricultural landscapes in the Brazilian Cerrado, Brazil's most important agribusiness region. We found that lizard abundance and observed richness were explained by percentage of agriculture cover while lizard diversity was predicted by canopy openness. However, snake abundance and alpha diversity were not affected by landscape or local vegetation structure. Species turnover was the main component explaining the beta diversity of lizards and snakes, and was solely attributed to variation in the number of trees across sampling sites. Contrary to our expectations, neither fragmentation measured via edge density nor landscape heterogeneity influenced Squamata's abundance and alpha diversity. Therefore, we reinforce and discuss how environmental and landscape changes can play a role in the distribution of species abundance and richness, and in structuring the patterns of species composition among reptile communities.

Influence of local vegetation and landscape structure on species abundance and alpha diversity

We found that lizards showed lower abundance and observed species richness in landscapes with higher percentages of agricultural cover. The conversion of natural vegetation into agricultural areas (e.g., crop and pasturelands) is a limiting factor of lizards' population size and species richness in several countries worldwide (Ribeiro et al. 2009; Ghosh and Basu 2020; López-Bedoya et al. 2022; Băncilă

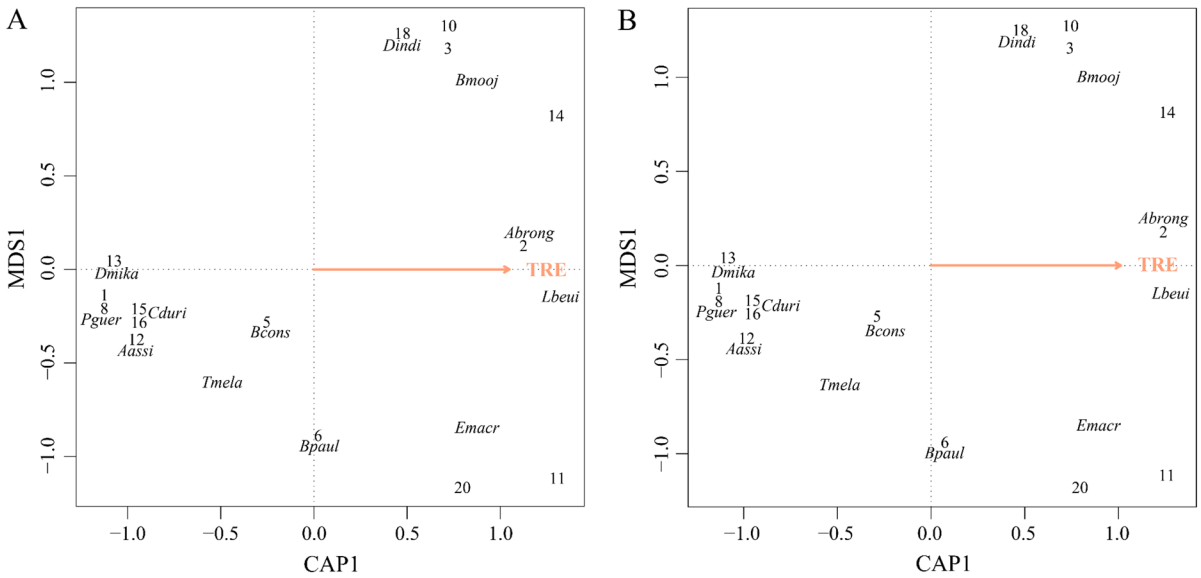


Fig. 6 Effect of number of trees (TRE) along the principal coordinate axis (CAP1) for the snake's communities beta diversity (**A**) and its turnover component (**B**), based on redundancy analysis (db-RDA). MDS1 represents the second axis that was not significant in our db-RDA analysis. Arrows indicate selected variables and the direction of effect. Abrong: *Amerotyphlops brongersmianus*; Aassi: *Apostolepis assimilis*; Bcons: *Boa constrictor*; Bmooj: *Bothrops moojeni*; Bpaul: *Bothrops pauloensis*; Cduri: *Crotalus durissus*; Dindi: *Dipsas indica*; Dmika: *Dipsas mikanii*; Emacr: *Erythrolamprus macrosomus*; Lbeui: *Liotyphlops beui*; Pguer: *Phimophis guerini*; Tmela: *Tantilla melanocephala*

et al. 2023), and sharp declines in reptile populations due to habitat loss has been reported across the globe (Jenkins et al. 2014; Palmeirim et al. 2017a; Falaschi et al. 2019; Doherty et al. 2020; Teixeira et al. 2022). This pattern is associated with reptiles' limited dispersal abilities (Jellinek et al. 2014; Guiller et al. 2022), sensibility to vegetation type and structure (Santos et al. 2016; Rotem and Ziv 2016; Rotem et al. 2020; Iglesias-Carrasco et al. 2022), and shape and size of remaining habitats (Jellinek et al. 2014; Ramírez et al. 2023). Our results support the predictions that lizard species are sensitive to habitat modifications. The conversion of natural habitats into agricultural systems directly affects reptile communities by reducing population sizes and favoring those species more resistant to altered environments, such as *Ameiva ameiva*, *T. torquatus*, *N. frenata*, and *C. modesta*.

A. ameiva is commonly found in disturbed areas throughout different Brazilian biomes (Vitt 1991; Sartorius et al. 1999; Benício et al. 2019; Jorge and Peña 2022), therefore, more tolerant to anthropogenic impact, possibly due to thermal constraints in more shaded areas (Sartorius et al. 1999) and a more resilient to impoverished habitat quality (Ledo and Colli 2016). *A. ameiva* commonly occurs in modified areas of forested environments where it finds cleared areas with higher temperatures. Likewise, the most abundant species we recorded in our study (*T. torquatus*, *N. frenata*, and *C. modesta*) were also frequently captured in Eucalyptus plantations adjacent to Cerrado *stricto sensu* in another study conducted in the central portion of the Cerrado ecoregion (Gainsburry and Colli 2014). This study also found that most Cerrado native species occurring in the area did not occur in the Eucalyptus plantation and were only captured in Cerrado *stricto sensu* suggesting species loss in areas where natural cover is no longer present. Our results corroborates with the findings that canopy openness is one of the main drivers of differences in species diversity with the majority species occurring in more open areas (Nogueira et al. 2005; Nogueira et al. 2005; Ramalho et al. 2021). In the Cerrado, changes in habitat structure are associated with changes in microclimate and suitability of the microhabitat used by lizards (Costa et al. 2020). Surprisingly, we did not find evidence that changes in canopy cover correlates

lizard abundance as reported in other studies (see Gainsburry and Colli 2014; Costa et al. 2020). We believe that such contrast can also be an artifact of the overall low lizard abundance in the study area and the shorter sampling period of our study.

The Brazilian Cerrado has only 11% of its total area formally protected, and a swath of natural vegetation has already been converted into crops and pasturelands (Rodrigues et al. 2022). The LAND-LTER landscape is predominantly occupied by intensive farming areas, and few and small areas of natural vegetation usually kept in the landscapes to meet the environmental law requirements (Santos et al. 2021). The loss of natural vegetation and landscape heterogeneity in LAND-LTER landscapes and increased patch shape irregularity led to decreased plant community diversity (Santos et al. 2021) and functional diversity (Martello et al. 2023). These changes can have high impacts on specialist species diversity, which depend on different habitat types to survive in different life stages (Băncilă et al. 2023; Silveira et al. 2023), including reptiles which are niche-specialized (Nogueira et al. 2011). On the other hand, a diversity of crops interposed with natural vegetation areas can offer a range of additional resources for species and benefit the biodiversity in agricultural landscapes (Sirami et al. 2019; Fulgence et al. 2022; Băncilă et al. 2023).

Agricultural landscape homogenization (Sirami et al. 2019) and the unsustainable management of agricultural areas (Santos et al. 2021) can cause several disturbs for reptiles (Romano et al. 2022; Mulhall et al. 2022; Băncilă et al. 2023; Ramírez et al. 2023). For example, croplands tend to increase habitat exposure to wind and water flux, reducing the efficiency of these areas to provide thermoregulatory habitats for reptiles (Ramírez et al. 2023). In Inselberg habitats in Australia, larger habitat patches had higher richness and diversity of lizards either when analyzed separately or even in a multivariate model that considered the effects of agricultural matrix and remnant vegetation (Michael et al. 2008). Also, lizards that have a preference for open habitats (Oliveira et al. 2021) can be severely affected by livestock activities due to cattle overgrazing, the replacement of native grasslands by exotic grasses (Oliveira et al. 2021; Băncilă et al. 2023), and the use of fire as a management practice (Oliveira et al. 2021; Mulhall et al. 2022). Terrestrial reptiles are particularly negatively affected in this scenario, especially those that depend on complex terrestrial structures that create thermal conditions for these species (Dorrough et al. 2012; Neilly et al. 2018). Heavy machinery for soil preparation commonly used in intensive farming can reduce microhabitats that serve as shelter, nests, foraging, and thermoregulation sites for reptiles (Mulhall et al. 2022; Ramírez et al. 2023).

Besides habitat loss, the use of agrichemicals adds to the biodiversity threats imposed by the expansion of agricultural cover at the expense of a reduction in natural vegetation cover. Reptiles are among the groups most affected by the use of pesticides (Todd et al. 2010). High pesticide loads are common practice in agricultural lands in Brazil, especially in soybean fields (Oliveira et al. 2021), the main crops grown in the studied landscape. The pesticides can be spread by spray drift or run-off to adjacent natural areas surrounding production fields and to water bodies, compromising habitats and consequently reducing the body condition of the reptiles, that have severe effects on the fitness of lizards (Mingo et al. 2017; Ramírez et al. 2023). For example, in urban environments, contamination by metals is the only predictor of reduced body condition of western tiger snakes *Notechis scutatus occidentalis*, among other health parameters such as the presence of parasites, scars, and tail loss (Lettoof et al. 2022). Adverse effects of habitat degradation on species' fitness-related traits have also been reported in lizards such as the *Crotaphytus collaris*, inhabiting the vegetation mosaics that form the Ozark glades in the United States. Lower fecundity of *C. collaris* populations is associated with habitat degradation and fragmentation (Brewster et al. 2018); both common processes associated with agricultural expansion worldwide and were reported in our study area for other taxa (Ribeiro et al. 2017; Ramalho et al. 2021; Silveira et al. 2023). Therefore, changes in habitat quality impose major threats to population dynamics by reducing fitness, consequently leading to a higher vulnerability of local extinction of Squamata populations and reducing biodiversity over time, and increasing the risk of ecosystem services loss. Our results suggest that this is the current situation of the most degraded sites in our study region. Therefore, to avoid local extinction and loss of ecosystem services provided by this group of organisms, it is paramount to develop multiscale

landscape management targeting the increase of habitat cover and strategies to diminish habitat anomalies derived from agricultural practices.

Reptiles perform several ecosystem functions that benefit humans, such as control of pest populations and disease vectors (Valencia-Aguilar et al. 2013). From this perspective, our result emphasizes that a continuous unsuitable expansion of agribusiness can threaten reptiles' survival and, consequently, the services they provide in the LAND LTER landscapes. Strips of natural vegetation tend to benefit some reptile species in Australian agricultural landscapes (Jellinek et al. 2014; Mulhall et al. 2022). The shape of these strips increases the area of contact with other ecosystems, providing additional resources (Ramírez et al. 2023), increasing landscape complementation (Sirami et al. 2019), connectivity, and a variety of habitats for reptile's thermoregulation (Ramírez et al. 2023). Therefore, restoration programs aiming at keeping habitat remnants and improving crop and habitat heterogeneity (Sirami et al. 2019; Băncilă et al. 2023) can favor the maintenance of reptiles and habitat specialists in the studied landscape.

Influence of local vegetation and landscape structure on beta diversity

Our findings show that dissimilarity between Squamata communities is high, and species turnover primarily drives such dissimilarities in the agricultural landscapes of the LAND LTER, corroborating our prediction. Likewise, more recent approaches have shown complex structuring in communities of different groups, including amphibians, lizards, snakes, and invertebrates (Larrea-Alcázar et al. 2011; Beirão et al. 2017; Mizsei et al. 2020; Ramalho et al. 2021). Most studies show that reptiles have a non-random community assembly, with high species turnover across the landscape in forests (e.g., Amazonia; Frazão et al. 2020) and open ecosystems like savanna (Tonial et al. 2012). Species turnover was previously identified as a main driver of reptiles' community dissimilarity in the Cerrado ecoregion,with a latitudinal spatial pattern in species replacement (Tonial et al. 2012). Here, we show that species turnover also shapes community dissimilarity at a narrower spatial scale.

Our findings suggest that the number of trees in the habitat patch is a local environmental filter determining lizard's and snake's community turnover since the number of trees explained the variation in Squamata's community beta diversity. Local environmental filters, such as vegetation structure (e.g., vegetation type, tree coverage, canopy cover), are also responsible for the dissimilarity in lizard communities in Amazonia (Peixoto et al. 2020) and Mexico (Peña-Joya et al. 2020). The Cerrado ecoregion is a mosaic formed by vegetation types (i.e., phytophysiognomies) whose characteristics largely differ from each other and generate a strong environmental gradient that ranges from open rocky grasslands to semideciduous and evergreen forests (Ribeiro and Walter 2008). Differences in vegetation structure (i.e., number of trees per area unit) are among the factors distinguishing these vegetational formations (Maracahipes-Santos et al. 2018) and driving compositional dissimilarity in other animal groups such as anurans (Thomé et al. 2021) in the Brazilian Cerrado. Variation in the number of trees in the Cerrado ecoregion can translate into variation in local habitat heterogeneity by adding horizontal and vertical stratification (Ribeiro and Walter 2008), expanding niche spaces for several organisms, including other plants, insects, birds, mammals, reptiles and small mammals (Crowther et al. 2015). Our results corroborate the limited number of studies assessing dissimilarity between Squamata reptile communities in the Cerrado (see Nogueira et al. 2005). Variations in reptile species composition in the Cerrado have been attributed to the type of vegetation, such as grasslands, savannas, and forests (e.g., Nogueira et al. 2005; Braga and Diniz 2015). Here, we highlight that the variation in the number of trees intrinsically associated with each Cerrado's phytophysiognomie plays a direct role in lizard and snake species turnover. Thus, it is likely that the effect of vegetation type found so far is an indirect effect of vegetation structure, particularly the number of trees, which filters species according to their niche requirements, constraining species' occurrence across the landscape (Vitt et al. 2003; Vitt and Pianka 2005). Most of the species recorded are restricted to different portions of the tree gradient (e.g., *Amphisbaena anaemariae, Copeoglossum nigropunctatum, Amphisbaena talisiae, Tropidurus itambere, Bachia bresslaui, Leotyphlops beui, Boa constrictus, Amerotyphlops brongersmianus*; Figs. 4 and 5) while only few species (e.g., *N. brasiliensis, T. torquatus,*

Erythrolamprus macrosomus; Figs. 4 and 5) occur along its whole extent. In addition, several snakes were exclusively found in environments with a low number of trees (*Phimophis guerini*, *Crotalus durissus*, *Tantilla melanocephala*, *Dipsas mikanii*, *D. indica*, *Apostolepis assimilis*, *Bothrops pauloensis*). Thus, the number of trees in the local environment is a crucial factor in determining the composition of Cerrado reptiles.

Even though we found no evidence that beta diversity and its turnover component are related to landscape structure, there are a handful of studies assessing the relationship between the composition of reptile communities with local and landscape variables in neotropical forested environments finding associations between them. For example, native vegetation cover in the landscape is the most important predictor to explain beta diversity between snake communities in undisturbed tropical forests (Fraga et al. 2018; Piatti et al. 2019). In disturbed dry forests of Mexico, Fraga-Ramirez et al. (2017) found that the patch density of riparian forest strongly influenced lizard composition, while snake composition also responded to the amount of riparian forest in the landscape and the density of dry forest patches. These findings reinforce the role of landscape composition and configuration in the structure of Squamata communities. In the Brazilian Cerrado, reptile species richness is often higher in open natural vegetation, such as savannas, than in forested environments, and most of these species are habitat specialists (Nogueira et al. 2009; Valdujo et al. 2009; Barros et al. 2022a, 2022b; Alexandre et al. 2023). Therefore, the effect of habitat loss can be more pronounced in endemic and specialist reptile species from savannas because open formations are more often replaced by crops and pasture than areas occupied by forested environments in the Brazilian Cerrado (Alencar et al. 2020). We suggest that further investigation of this concept should be conducted across the Cerrado ecoregion. A multivariate modelling approach is also needed to investigate interactions between landscape predictors and exclude confounding effects that our univariate models may fail to capture. We also highlight that savannas and grasslands in the Cerrado ecoregion should be prioritized in conservation planning, especially because of the replacement of species across the vegetation structure gradient.

Conclusion

Understanding the effects of crops on biodiversity is essential to help design effective environmental policies for biodiversity conservation and food production. Here, we show that the amount of agricultural cover and vegetation structure shapes Squamata reptile diversity and distribution in agricultural landscapes in the Brazilian Cerrado. Landscapes dominated by monocultures negatively affected species richness and abundance on a scale between 400 and 600 m. These findings suggest that communities and populations get smaller in areas with low habitat availability, consequently increasing the chances of local extinction. Beta diversity between landscapes revealed that lizards and snake species inhabiting open areas, i.e., environments with fewer trees, are generally replaced by others in forested vegetation. The Brazilian Cerrado is the most important agribusiness region in Brazil, leading to the loss of more than 50% of its natural vegetation cover in the last 60 years, the highest rate of land cover change among all Brazilian ecoregions, predominantly presenting small fragments with less than < 1 ha (Strassburg et al. 2017; Alencar et al. 2020; Pompeu et al. 2024). Our findings suggest that the conservation of more forests and savannas in the Cerrado landscapes should provide more suitable environments, increasing Squamata diversity.

In the LAND LTER, similar to most landscapes in Cerrado, landscape mosaics are comprised of massive swaths of intensive agriculture and few and thin strips of native cover vegetation, which are preserved to comply with the 20% minimum area of natural vegetation required by environmental law (Santos et al. 2021). The current arrangement and composition of the landscapes in the Cerrado may compromise the conservation of Squamata because the efficiency of the remaining forests and savanna patches in conserving community diversity depends on their structure and quality. Thus, incentives to maintain or restore native vegetation in the LAND LTER is of utmost importance to the conservation of Squamata species at the landscape level.

Acknowledgements GSM received a master fellowship from Fundação de Amparo à Pesquisa do Estado de Goiás (FAPEG, process no. 2017.1026.7000.599) and a PhD fellowship from Coordenação de Aperfeiçoamento de Pessoal de Nível Superior (CAPES, process no. 88882.386630/2019-01). WPR is a

regional development researcher supported by FAPEG and the Conselho Nacional de Desenvolvimento Científico e Tecnológico (CNPq, process no. 317724/2021-5). JSS received a CAPES postdoctoral fellowship (Process no. 2019/09713-6) and a postdoctoral grant from the Fundação de Amparo à Pesquisa do Estado de São Paulo (FAPESP, process no. 2022/00166-5). RGC, MCR and NMM have continuously been supported by productivity grants from Conselho Nacional de Desenvolvimento Científico e Tecnológico (CNPq) which we gratefully acknowledge. We thank private landholders for access to their properties. We also thank all the colleagues for assistance with field work: Alejandro Valencia, Ana Carolina Moura, Ana Paula Pio, Denes Ferraz, Elvimar Gomes, Gabrielly Rodrigues, Gessileny Rosa, James Fagno, Jéssica Ribeiro, José Ulisses Araújo, Jourdan Calil, Kalliman Paiva, Leonardo Marques, Lhayza Cruvinel, Lucas Gaehwiler, Matheus Araújo, Nayala Ferreira, Nicole Fighera, Philip Teles and Rafaela Macedo (in memoriam). Field work was performed under the license given by the Ministério do Meio Ambiente/Instituto Chico Mendes de Conservação da Biodiversidade (no. 58600-1).

Author contributions Conceptualization: RGC (equal), MCR (equal), WVS (supporting). Collected the data: GSM (equal), WPR (equal), IBF (equal). Funding acquisition: RGC. Formal analysis: GSM (equal), WPR (equal), PS (equal). Generated land cover map: JSS (lead), MCR (supporting). Prepared figures and tables: GSM. Project administration: RGC (lead), MCR (supporting), JSS (supporting). Writing–original draft: GSM (equal), WPR (equal), PS (equal). Writing–review and editing: GSM (equal), WPR (equal), PS (equal), RGC (equal), JSS (equal), NMM (supporting), WVS (supporting).

Funding This work was supported by grants given to the LAND-LTER research network supported by MCT/CNPq/CAPES (project no. 441278/2016–7) and CNPq (no. 445419/2024–5), FAPEG (project no. 201710267000331 and 202010267000404), CAPES/PROCAD (project no. 88881.068425/2014–01). PPBio Biota Cerrado funded by CNPq (project no. 441166/2023–7).

Data availability Additional data with the sampling locations and detailed results is provided within the supplementary information file.

Declarations

Competing interests The authors declare no competing interests.

Open Access This article is licensed under a Creative Commons Attribution-NonCommercial-NoDerivatives 4.0 International License, which permits any non-commercial use, sharing, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if you modified the licensed material. You do not have permission under this licence to share adapted material derived from this article or parts of it. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit http://creativecommons.org/licenses/by-nc-nd/4.0/.

References

Alencar A, Shimbo ZJ, Lenti F et al (2020) Mapping three decades of changes in the Brazilian Savanna native vegetation using landsat data processed in the google earth engine platform. Remote Sens 12:924.

Alexandre RJR, Bergamini FM, Spigoloni ZA, Dias-Silva K, Vieira RRS, Guerra V, Bastos RP, Vieira TB (2023) Squamate reptiles as indicators in fragments of Brazilian cerrado. Stud Neotrop Fauna Environ. https://doi.org/10.1080/01650521.2023.2276228

Alroy J (2015) Current extinction rates of reptiles and amphibians. Proc Natl Acad Sci USA 112(42):13003–13008.

Arita HT, Rodríguez P (2002) Geographic range, turnover rate and the scaling of species diversity. Ecography 25:541–550.

Azevedo JAR, Valdujo PH, Nogueira CC (2016) Biogeography of anurans and squamates in the Cerrado hotspot: coincident endemism patterns in the richest and most impacted savanna on the globe. J Biogeogr 43:2454–2464.

Băncilă RI, Lattuada M, Sillero N (2023) Distribution of amphibians and reptiles in agricultural landscape across Europe. Landsc Ecol 38:861–874.

Banks-Leite C, Ewers RM, Folkard-Tapp H, Fraser A (2020) Countering the effects of habitat loss, fragmentation, and degradation through habitat restoration. One Earth 3:672–676.

Barros RA, Dorado-Rodrigues TF, Strüssmann C (2022a) Taxonomic, functional, and phylogenetic diversity of lizard assemblages across habitats and seasons in a Brazilian Cerrado area. Austral Ecol 47:983–996.

Barros RA, Dorado-Rodrigues TF, Valadão RM, Strüssmann C (2022b) Diversity patterns of lizard assemblages from a protected habitat mosaic in the Brazilian Cerrado savanna. J Trop Ecol 38:340–350.

Baselga A (2010) Partitioning the turnover and nestedness components of beta diversity. Glob Ecol Biogeogr 19:134–143.

Baselga A (2013) Separating the two components of abundance-based dissimilarity: balanced changes in abundance vs. abundance gradients. Methods Ecol Evol 4:552–557.

Beca G, Vancine MH, Carvalho CS et al (2017) High mammal species turnover in forest patches immersed in biofuel plantation. Biol Conserv 210:352–359.

Beirão MV, Neves FS, Penz CM et al (2017) High butterfly beta diversity between Brazilian cerrado and cerrado–caatinga transition zones. J Insect Conserv 21:849–860.

Benício RA, Ortega Z, Mencia A, Passos DC (2019) Microhabitat selection of Ameiva ameiva (Linnaeus, 1758), in the Brazilian Pantanal. Herpetozoa 31:211–218

Ben-Shachar MS, Lüdecke D, Makowski D (2020) Effectsize: estimation of effect size indices and standardized parameters. J Open Source Softw 5(56):2815.

Bjornstad ON (2022). ncf: spatial covariance functions. R package version 1.3–2, https://github.com/objornstad/ncf.

Böhm M, Collen B, Baillie JEM et al (2013) The conservation status of the world's reptiles. Biol Conserv 157:372–385.

Bolker BM (2008) Ecological models and data in R. Princeton University Press, Princeton

Braga L, Diniz IR (2015) Importance of habitat heterogeneity in richness and diversity of moths (Lepidoptera) in Brazilian Savanna. Environ Entomol 44(3):499–508.

Brewster CL, Beaupre SJ, Willson JD (2018) Habitat loss and local extinction: linking population declines of eastern collared lizards (Crotaphytus collaris) to habitat degradation in Ozark glades. J Herpetol 52(3):352–360.

Buckley LB, Jetz W (2008) Linking global turnover of species and environments. Proc Natl Acad Sci USA 105(46):17836–17841.

Burnham KP, Anderson D (2002) Model selection and multimodel inference: a practical information-theoretic approach, 2nd edn. Springer, New York

Cechin SZ, Martins MRC (2000) Eficiência de armadilhas de queda (pitfall traps) em amostragens de anfíbios e répteis no Brasil. Rev Bras Zool 17(3):729–740.

Chao A, Kubota Y, Zelený D, Chiu C-H et al (2020) Quantifying sample completeness and comparing diversities among assemblages. Ecol Res 35:292–314.

Chao A, Hu K-H (2024) The iNEXT.4steps package: Four-Step Biodiversity Analysis based on iNEXT. R package available from CRAN.

Cohen J, Cohen P, West SG, Aiken LS (2003) Applied multiple regression/correlation analysis for the behavioral sciences, 3rd edn. Lawrence Erlbaum Associates Publishers, Mahwah

Colli GR, Bastos RP, Araújo 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

Costa BM, Pantoja DL, Sousa HC et al (2020) Long-term, fire-induced changes in habitat structure and microclimate affect Cerrado lizard communities. Biodivers Conserv 29:1659–1681.

Cox N, Young BE, Bowles P, Fernandez M, Marin J, Rapacciuolo G et al (2022) A global reptile assessment highlights shared conservation needs of tetrapods. Nature 605(7909):285–290.

Crowther T, Glick H, Covey K et al (2015) Mapping tree density at a global scale. Nature 525:201–205.

D'Cruze N, Kumar S (2011) Effects of anthropogenic activities on lizard communities in northern Madagascar. Anim Conserv 14:542–552.

Dobrovolski R, Melo AS, Cassemiro FAS, Diniz-Filho JAF (2012) Climatic history and dispersal ability explain the relative importance of turnover and nestedness components of beta diversity. Glob Ecol Biogeogr 21:191–197.

Doherty TS, Balouch S, Bell K et al (2020) Reptile responses to anthropogenic habitat modification: a global meta-analysis. Glob Ecol Biogeogr 29:1265–1279.

Dorrough J, Mcintyre S, Brown G, Stol J, Barrett G, Brown A (2012) Differential responses of plants, reptiles and birds to grazing management, fertilizer and tree clearing. Austral Ecol 37(5):569–582.

Dray S, Bauman D, Blanchet G et al (2023) Adespatial: Multivariate Multiscale Spatial Analysis. R package version 0.3–21, https://CRAN.R-project.org

Dvořáková D, Šipoš J, Suchomel J (2023) Impact of agricultural landscape structure on the patterns of bird species diversity at a regional scale. Avian Res 14:100147.

Falaschi M, Manenti R, Thuiller W, Ficetola GF (2019) Continental-scale determinants of population trends in European amphibians and reptiles. Glob Chang Biol 25:3504–3515.

Fischer J, Lindenmayer DB (2007) Landscape modification and habitat fragmentation: a synthesis. Glob Ecol Biogeogr 16:265–280.

Foley JA, DeFries R, Asner GP et al (2005) Global consequences of land use. Science 309:570–574.

Fraga R, Ferrão M, Stow AJ, Magnusson WE, Lima AP (2018) Different environmental gradients affect different measures of snake β-diversity in the Amazon rainforests. PeerJ 6:e5628.

Fraga-Ramírez Y, Suazo-Ortuño I, Avila-Cabadilla LD, Alvarez-Anorve M, Alvarado-Diaz J (2017) Multiscale analysis of factors influencing herpetofaunal assemblages in early successional stages of a tropical dry forest in western Mexico. Biol Conserv 209:196–210.

Frazão L, Oliveira ME, Menin M, Campos J, Almeida A, Kaefer IL, Hrbek T (2020) Species richness and composition of snake assemblages in poorly accessible areas in the Brazilian Amazonia. Biota Neotrop 20(1):e20180661.

Fulgence TR, Martin DA, Randriamanantena R (2022) Differential responses of amphibians and reptiles to land-use change in the biodiversity hotspot of north-eastern Madagascar. Anim Conserv 25(4):492–507

Gainsbury AM, Colli GR (2014) Effects of abandoned Eucalyptus plantations on lizard communities in the Brazilian Cerrado. Biodivers Conserv 23:3155–3170.

Garden JG, Mcalpine CA, Possingham HP, Jones DN (2007) Habitat structure is more important than vegetation composition for local-level management of native terrestrial reptile and small mammal species living in urban remnants: a case study from Brisbane, Australia. Austral Ecol 32(6):669–685.

Ghosh D, Basu P (2020) Factors influencing herpetofauna abundance and diversity in a tropical agricultural landscape mosaic. Biotropica 52:927–937.

González C, Macip-Ríos R, Suazo-Ortuño I (2022) Phylogenetic structure and diversity among herpetofaunal communities along a successional gradient of a tropical dry forest in Mexico. Perspect Ecol Conserv 20(3):249–255.

Gotelli NJ (2000) Null model analysis of species co-occurrence patterns. Ecology 81(9):2606–2621.

Gotelli NJ, McCabe DJ (2002) Species co-occurrence: a meta-analysis of J. M. Diamond's assembly rules model. Ecology 83:2091.

Guedes TB, Entiauspe-Neto OM, Costa HC (2023) Lista de répteis do Brasil: atualização de 2022. https://doi.org/10.5281/zenodo.7829013

Guiller G, Legentilhomme J, Boissinot A et al (2022) Response of farmland reptiles to agricultural intensification: collapse of the common adder Vipera berus and the western green lizard Lacerta bilineata in a hedgerow landscape. Anim Conserv 25:849–864.

Haddad NM, Brudvig LA, Clobert J et al (2015) Habitat fragmentation and its lasting impact on Earth's ecosystems. Sci Adv 1:e1500052.

Hesselbarth MHK, Sciaini M, With KA et al (2019) landscapemetrics: an open-source R tool to calculate landscape metrics. Ecography 42:1648–1657.

Holland AF, Sanger DM, Gawle CP et al (2004) Linkages between tidal creek ecosystems and the landscape and demographic attributes of their watersheds. J Exp Mar Bio Ecol 298(2):151–178.

Huais PY (2018) Multifit: an R function for multi-scale analysis in landscape ecology. Landsc Ecol 33:1023–1028.

Iglesias-Carrasco M, Medina I, Ord TJ (2022) Global effects of forest modification on herpetofauna communities. Conserv Biol 37:e13998.

Jackson HB, Fahrig L (2015) Are ecologists conducting research at the optimal scale? Global Ecol Biogeogr 24:52–63.

Jellinek S, Parris KM, McCarthy MA, Wintle BA, Driscoll DA (2014) Reptiles in restored agricultural landscapes: the value of linear strips, patches and habitat condition. Anim Conserv 17(6):544–554.

Jenkins RKB, Tognelli MF, Bowles P et al (2014) Extinction risks and the conservation of Madagascar's reptiles. PLoS ONE 9(8):e100173.

Jorge RF, Pena AP (2022) Reptiles from an Urban Park in the Cerrado of Central Brazil. Herpetol Conserv Biol 17(2):413–422

Kanowski JJ, Reis TM, Catterall CP, Piper SD (2006) Factors affecting the use of reforested sites by reptiles in cleared rainforest landscapes in tropical and subtropical Australia. Restor Ecol 14(1):67–76.

Kay GM, Tulloch A, Barton PS, Cunningham SA, Driscoll DA, Lindenmayer DB (2018) Species co-occurrence networks show reptile community reorganization under agricultural transformation. Ecography 41(1):113–125.

Kearney M, Shine R, Porter WP (2009) The potential for behavioral thermoregulation to buffer "cold-blooded" animals against climate warming. Proc Natl Acad Sci USA 106:3835–3840.

Keinath DA, Doak DF, Hodges KE et al (2017) A global analysis of traits predicting species sensitivity to habitat fragmentation. Glob Ecol Biogeogr 26:115–127.

Kremen C, Iles A, Bacon C (2012) Diversified farming systems: an agroecological, systems-based alternative to modern industrial agriculture. Ecol Soc 17(4):44.

Larrea-Alcázar DM, Embert D, Aguirre LF et al (2011) Spatial patterns of biological diversity in a neotropical lowland savanna of northeastern Bolivia. Biodivers Conserv 20:1167–1182.

Ledo RMD, Colli GR (2016) Silent death: the new Brazilian forest code does not protect lizard assemblages in Cerrado Riparian Forests. South Am J Herpetol 11(2):98–109.

Lettoof DC, Cornelis J, Jolly CJ, Aubret F, Gagnon MM, Hyndman TH, Barton DP, Bateman PW (2022) Metal(loid) pollution, not urbanisation nor parasites predicts low body condition in a wetland bioindicator snake. Environ Pollut 295:118674.

Longrich NR, Vinther J, Pyron RA et al (2015) Biogeography of worm lizards (amphisbaenia) driven by end-Cretaceous mass extinction. Proc Royal Soc b: Biol Sci 282(1806):1–10.

López-Bedoya PA, Cardona-Galvis EA, Urbina-Cardona JN, Edwards FA, Edwards DP (2022) Impacts of pastures and forestry plantations on herpetofauna: a global meta-analysis. J Appl Ecol 59:3038–3048.

Lüdecke D, Ben-Shachar M, Patil I et al (2021) Performance: an R package for assessment, comparison and testing of statistical models. J Open Source Softw 6:3139.

Luedtke JA, Chanson J, Neam K, Hobin L, Maciel AO, Catenazzi A et al (2023) Ongoing declines for the world's amphibians in the face of emerging threats. Nature 622(7982):308–314.

MapBiomas (2022) Collection of Brazilian land cover and use map annual seriesv.6.0. https://mapbiomas.org.

Maracahipes-Santos L, Santos JOD, Reis SM, Lenza E (2018) Temporal changes in species composition, diversity, and woody vegetation structure of savannas in the Cerrado-Amazon transition zone. Acta Bot Bras 32:254–263.

Martello F, Santos JS, Silva-Neto CM, Cássia-Silva C, Siqueira KN, Ataíde MVRD, Ribeiro MC, Collevatti RG (2023) Landscape structure shapes the diversity of plant reproductive traits in agricultural landscapes in the Brazilian Cerrado. Agric Ecosyst Environ 341:108216.

Martins M, Oliveira ME (1998) Natural history of snakes in forests of the Manaus region, Central Amazonia. Brazil Herpetol Nat Hist 6(2):78–150

Martins TO, Silva-Neto CM, Siqueira KN et al (2021) Accumulated litter and nutrient stock in biomass and in soil in forest formations in the Cerrado. Sci for 49(129):e3427.

McGarigal K, Cushman SA, Ene E (2012) FRAGSTATS v4: spatial pattern analysis program for categorical and continuous maps. Computer software program produced by the authors at the University of Massachusetts. University of Massachusetts, Amherst

Michael DR, Cunningham RB, Lindenmayer DB (2008) A forgotten habitat? Granite inselbergs conserve reptile diversity in fragmented agricultural landscapes. J Appl Ecol 45:1742–1752.

Mingo V, Lötters S, Wagner N (2017) The impact of land use intensity and associated pesticide applications on fitness and enzymatic activity in reptiles - a field study. Sci Total Environ 590–591:114–124.

Mizsei E, Fejes Z, Malatinszky Á et al (2020) Reptile responses to vegetation structure in a grassland restored for an endangered snake. Community Ecol 21:203–212.

Moreno CE, Calderón-Patrón JM, Martín-Regalado N et al (2018) Measuring species diversity in the tropics: a review of methodological approaches and framework for future studies. Biotropica 50(6):929–941.

Moura MR, Costa HC, Argôlo AJS, Jetz W (2017) Environmental constraints on the compositional and phylogenetic beta-diversity of tropical forest snake assemblages. J Anim Ecol 86:1192–1204.

Mulhall SJ, Sitters H, Di Stefano J (2022) Vegetation cover and configuration drive reptile species distributions in a fragmented landscape. Wildl Res. https://doi.org/10.1071/WR21063

Munstermann MJ, Heim NA, McCauley DJ, Payne JL, Upham NS, Wang SC, Knope ML (2022) A global ecological signal of extinction risk in terrestrial vertebrates. Conserv Biol 36(3):e13852.

Myers N, Mittermeier RA, Mittermeier CG et al (2000) Biodiversity hotspots for conservation priorities. Nature 403:853–858.

Neilly H, Nordberg EJ, VanDerWal J, Schwarzkopf L (2018) Arboreality increases reptile community resistance to disturbance from livestock grazing. J Appl Ecol 55(2):786–799.

Newbold T, Hudson LN, Hill SLL et al (2015) Global effects of land use on local terrestrial biodiversity. Nature 520:45–50.

Newbold T, Hudson LN, Arnell AP et al (2016) Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment. Science 353:288–291.

Nogueira C, Valdujo PH, França FGR (2005) Habitat variation and lizard diversity in a Cerrado area of Central Brazil. Stud Neotrop Fauna Environ 40:105–112.

Nogueira C, Colli GR, Martins M (2009) Local richness and distribution of the lizard fauna in natural habitat mosaics of the Brazilian Cerrado. Austral Ecol 34:83–96.

Nogueira C, Ribeiro S, Costa GC, Colli GR (2011) Vicariance and endemism in a Neotropical savanna hotspot: distribution patterns of Cerrado squamate reptiles. J Biogeogr 38:1907–1922.

Nogueira CC, Argôlo AJS, Arzamendia V et al (2019) Atlas of Brazilian snakes: verified point-locality maps to mitigate the Wallacean shortfall in a megadiverse snake Fauna. South Am J Herpetol 14:S1–S274.

Oksanen J, Blanchet FG, Friendly M et al (2020) Vegan: community ecology. Package. R package version 2.5–7

Oliveira MM, Morato RG, Jorge RSP, Paula RC (2021) Agricultural activities and threat to fauna in Brazil: an analysis of the red book of endangered Brazilian Fauna. Pap Avulsos Zool 61:e20216193.

Palmeirim AF, Vieira MV, Peres CA (2017a) Herpetofaunal responses to anthropogenic forest habitat modification across the neotropics: insights from partitioning β-diversity. Biodivers Conserv 26:2877–2891.

Palmeirim AF, Vieira MV, Peres CA (2017b) Non-random lizard extinctions in land-bridge Amazonian forest islands after 28 years of isolation. Biol Conserv 214:55–65.

Peixoto GM, Fraga R, Araújo MC et al (2020) Hierarchical effects of historical and environmental factors on lizard assemblages in the upper Madeira River. Brazilian Amazonia Plos One 15(6):e0233881.

Peña-Joya KE, Cupul-Magaña FG, Rodríguez-Zaragoza FA et al (2020) Spatio-temporal discrepancies in lizard species and functional diversity. Community Ecol 21:1–12.

Pereira I, Calil F, Martins T et al (2016) Fire efect on the seasonal forest structure in the Cerrado biome. Floresta 46(4):499.

Piatti L, Rosauer DF, Nogueira CC et al (2019) Snake diversity in floodplains of central South America: is flood pulse the principal driver? Acta Oecol 97:34–41.

Pompeu J, Assis TO, Ometto JP (2024) Landscape changes in the Cerrado: challenges of land clearing, fragmentation and land tenure for biological conservation. Sci Total Environ 906:167581.

Püttker T, Crouzeilles R, Almeida-Gomes M et al (2020) Indirect effects of habitat loss via habitat fragmentation: a cross-taxa analysis of forest-dependent species. Biol Conserv 241:108368.

Pyron RA (2017) Novel approaches for phylogenetic inference from morphological data and total-evidence dating in squamate reptiles (Lizards, Snakes, and Amphisbaenians). Syst Biol 66:38–56.

Pyron RA, Burbrink FT, Wiens JJ (2013) A phylogeny and revised classification of Squamata, including 4161 species of lizards and snakes. BMC Evol Biol 13:93.

QGIS Development Team (2017) QGIS Geographic Information System. Open Source Geospatial Foundation Project. http://qgis.osgeo.org

R Core Team (2022). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org.

R Core Team (2024). R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. Available from https://www.R-project.org.

Ramalho WP, Prado VHM, Signorelli L, With KA (2021) Multiple environmental filters and competition affect the spatial co-occurrence of pond-breeding anurans at both local and landscape scales in the Brazilian Cerrado. Landsc Ecol 36:1663–1683.

Ramalho WP, With KA, Mesquita GS, Arruda FV, Guerra V, Ferraz D, Andrade MS, Prado VHM (2022) Habitat fragmentation rather than habitat amount or habitat split reduces the diversity and abundance of ground-dwelling anurans within forest remnants of the Brazilian Cerrado. J Nat Conserv 69:126259.

Ramírez AMP (2023) Effects of landscape structure and patch characteristics on the density of central populations of the eastern green lizard Lacerta viridis. Ecol Evol 13(8):e10419.

Rausch LL, Gibbs HK, Schelly I et al (2019) Soy expansion in Brazil's Cerrado. Conserv Lett 12:e12671.

Regolin AL, Ribeiro MC, Martello F et al (2020) Spatial heterogeneity and habitat configuration overcome habitat composition influences on alpha and beta mammal diversity. Biotropica 52:969–980.

Ribeiro JF, Walter BMT (2008) As principais fitofisionomias do Bioma Cerrado. In: Almeida SP, Ribeiro JF (eds) Sano SM. Cerrado Ecologia e Flora, Planaltina, pp 152–212

Ribeiro R, Santos X, Sillero N, Carretero MA, Llorente GA (2009) Biodiversity and land uses at a regional scale: Is agriculture the biggest threat for reptile assemblages? Acta Oecol 35(2):327–334.

Ribeiro J, Colli GR, Batista R, Soares A (2017) Landscape and local correlates with anuran taxonomic, functional and phylogenetic diversity in rice crops. Landsc Ecol 32:1599–1612.

Rodrigues AA, Macedo MN, Silvério DV, Maracahipes L, Coe MT, Brando PM, Shimbo JZ, Rajão R, Soares-Filho B, Bustamante MMC (2022) Cerrado

deforestation threatens regional climate and water availability for agriculture and ecosystems. Glob Chang Biol 28(22):6807–6822.
Romano A, Rosa G, Salvidio S, Novaga R, Costa A (2022) How landscape and biotic interactions shape a Mediterranean reptile community. Landsc Ecol 37:2915–2927.
Rotem G, Ziv Y (2016) Crop diversity and rotation may increase dispersal opportunities of reptiles in a heterogeneous agroecosystem. Agric Ecosyst Environ 235:32–37.
Rotem G, Giladi I, Bouskila A, Ziv Y (2020) Scale-dependent correlates of reptile communities in natural patches within a fragmented agroecosystem. Landsc Ecol 35:2339–2355.
Santos X, Badiane A, Matos C (2016) Contrasts in short- and long-term responses of Mediterranean reptile species to fire and habitat structure. Oecologia 180:205–216.
Santos JS, Dodonov P, Oshima JEF et al (2021) Landscape ecology in the Anthropocene: an overview for integrating agroecosystems and biodiversity conservation. Perspect Ecol Conserv 19:21–32.
Sartorius SS, Vitt LJ, Colli GR (1999) Use of naturally and anthropogenically disturbed habitats in Amazonian rainforest by the teiid lizard Ameiva ameiva. Biol Conserv 90(2):91–101.
Scott DM, Brown D, Mahood S, Denton B, Silburn A, Rakotondraparany F (2006) The impacts of forest clearance on lizard, small mammal and bird communities in the arid spiny forest, southern Madagascar. Biol Conserv 127:72–87.
Silva DJ, Palmeirim AF, Santos-Filho M, Sanaiotti TM, Peres CA (2022) Habitat quality, not patch size, modulates lizard responses to habitat loss and fragmentation in the Southwestern Amazon. J Herpetol 56(1):75–83.
Silveira P, Iop S, Santos JS et al (2023) Amount and isolation of aquatic habitat drive anuran diversity in agricultural landscapes in the Brazilian Cerrado. Landsc Ecol 38:2261–2275.
Sirami C, Gross N, Baillod AB et al (2019) Increasing crop heterogeneity enhances multitrophic diversity across agricultural regions. Proc Nat Acad Sci 116:16442–16447.
Sousa FG, Santos JS, Martello F et al (2022) Natural habitat cover and fragmentation per se influence orchid-bee species richness in agricultural landscapes in the Brazilian Cerrado. Apidologie 53:20.
Strassburg BBN, Brooks T, Feltran-Barbieri R et al (2017) Moment of truth for the Cerrado hotspot. Nat Ecol Evol 1:0099.
Tan WC, Herrel A, Rödder D (2023) A global analysis of habitat fragmentation research in reptiles and amphibians: what have we done so far? Biodivers Conserv 32:439–468.
Teixeira CC, Trevelin LC, Santos-Costa MC et al (2022) Synergistic effects of climate and landscape change on the conservation of Amazonian lizards. PeerJ 10:e13028.
Teixido AL, Gonçalves SRA, Fernández-Arellano GJ et al (2020) Major biases and knowledge gaps on fragmentation research in Brazil: implications for conservation. Biol Conserv 251:108749.
Teixido AL, Sehn H, Quintanilla LG et al (2021) A meta-analysis of the effects of fragmentation on the megadiverse herpetofauna of Brazil. Biotropica 53:726–737.
Thioulouse J, Dray S, Dufour A, Siberchicot A, Jombart T, Pavoine S (2018) Multivariate analysis of ecological data with ade4. Springer, Cham
Thomé MTC, Martins M, Brasileiro CA (2021) Higher diversity in open Cerrado supports the role of regional processes in shaping an anuran assemblage in southeastern Brazil. South Am J Herpetol 19:1–7.
Todd BD, Willson JD, Gibbons JW (2010) The global status of reptiles and causes of their decline. In: Sparling DW, Linder G, Bishop CA, Krest S (eds) Ecotoxicology of amphibians and reptiles, 2nd edn. CRC Press, Boca Raton, pp 47–67
Tonial MLS, Silva HLR, Tonial IJ et al (2012) Geographical patterns and partition of turnover and richness components of beta-diversity in faunas from Tocantins river valley. Braz J Biol 72(3):497–504.
Valdujo PH, Nogueira CC, Baumgarten L et al (2009) Squamate reptiles from Parque Nacional das Emas and surroundings, Cerrado of Central Brazil. Check List 5(3):405–417.
Valencia-Aguilar A, Cortés-Gómez AM, Ruiz-Agudelo CA (2013) Ecosystem services provided by amphibians and reptiles in Neotropical ecosystems. Int J Biodivers Sci Ecosyst Serv Manag 9(3):257–272.
Viana DS, Figuerola J, Schwenk K et al (2016) Assembly mechanisms determining high species turnover in aquatic communities over regional and continental scales. Ecography 39:281–288.
Vitt LJ (1991) An introduction to the ecology of Cerrado lizards. J Herpetol 25(1):79–90
Vitt LJ, Pianka ER (2005) Deep history impacts present-day ecology and biodiversity. Proc Natl Acad Sci 102(22):7877–7881.
Vitt LJ, Pianka ER, Cooper WE Jr, Schwenk K (2003) History and the global ecology of squamate reptiles. Am Nat 162(1):44–60.
Whittaker RH (1972) Evolution and measurement of species diversity. Taxon 21:213–251.
Winemiller KO, Pianka ER (1990) Organization in natural assemblages of desert lizards and tropical fishes. Ecol Monogr 60(1):27–55.
With KA (2019) Essentials of Landscape Ecology. Oxford University Press, New York
Zuur AF, Ieno EN, Elphick CS (2010) A protocol for data exploration to avoid common statistical problems. Methods Ecol Evol 1:3–14.

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.