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Effects of changes in the riparian forest on the butterfly community (Insecta: Lepidoptera) in Cerrado areas

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ABSTRACT

Preserved riparian vegetation usually has greater environmental complexity than the riparian vegetation modified by human actions. These systems may have a greater availability and diversity of food resources for the species. Our objective was to evaluate the effect of changes on the structure of the riparian forest on species richness, beta diversity and composition of butterfly species in the Cerrado of Mato Grosso. We tested the hypotheses that: (i) higher species richness and (ii) beta diversity would be recorded in more preserved environments; and (iii) species composition would be more homogeneous in disturbed habitats. For hypothesis testing, the riparian vegetation of eight streams were sampled in four periods of the year in a fixed transect of 100 m along the shores. The richness of butterfly species is lower in disturbed than in preserved areas. However, species richness is not affected by habitat integrity. Beta diversity differed among sites, such that preserved sites have greater beta diversity, showing greater variation in species composition. In addition, beta diversity was positively affected by environmental heterogeneity. A total of 23 of the 84 species sampled occurred only in the changed environment, 42 were exclusive to preserved sites and 19 occurred in both environments. The environmental change caused by riparian forest removal drastically affects the butterfly community. Therefore, riparian vegetation is extremely important for butterfly preservation in the Cerrado and may be a true biodiversity oasis, especially during the dry periods, when the biome undergoes water stress and resource supply is more limited.

Introduction

Riparian forest is defined as the forest vegetation established along the shores of medium to large rivers. This vegetation has important features and resources for the survival of animal communities (Ribeiro and Walter, 1998, 2001), providing high humidity, temperatures that are cooler and more stable than that of the surrounding landscape, low light incidence and abundant food resources (Brown, 2000). In addition, riparian forests are also ecological corridors for wildlife, connect different habitat fragments and effectively increase percolation in the landscape scale (Metzger, 2010). The density of animal species in the riparian forests in dryer regions and seasons may lead to an intense biotic pressure among the populations present, leading to genetic diversification and evolution, and consequently to a high species richness (Brown, 2000).

Natural habitats have been drastically reduced and/or modified by intense human activities, which have caused only few vegetation remnants to persist. The primary cause of the decline of species diversity in riparian forest is habitat loss. Forest fragmentation increases with the loss of original habitat and has its size reduced, consequently, the isolation of habitat patches increases. Changes in the use of the riparian vegetation, as well as the variation in the intensity of use may cause the loss of some specific environmental conditions or resources that are important for the species. Such loss may thus reduce abundance, cause local extinction of more sensitive species or in some cases favor the input and the establishment of generalist or invasive species (Barrella et al., 2000).

The Cerrado is one of the Brazilian biomes that have most suffered the effects of anthropization. This biome has high levels of endemism (Rodrigues, 2005; Camargo, 2001; Pinheiro, 2008) for several groups, in addition a high species diversity. An example is the butterfly fauna, which consists on a group with high species richness, extremely dependent on specific resources (plants) and highly faithful to microhabitats (Brown and Freitas, 2002; Freitas

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et al., 2003). Butterflies are especially dependent on specific resources associated with environments with high humidity and abundant food resources (Brown, 1992, 2000; Camargo, 2001).

Butterflies respond quickly to environmental and climatic changes are relatively easy to monitor, and its community structure is easily assessed (Brown, 1992; Brown and Freitas, 1999, 2000; Raimundo et al., 2003; Uehara-Prado et al., 2009). These insects are involved in many interactions in the surrounding environment, for example, pollination and predation (Bogiani et al., 2012). Therefore, butterflies are important for the functioning of ecosystem services and are good models for ecological studies. Given this scenario, our objective was to evaluate the effect of the riparian forest changes on species richness, beta diversity and composition of butterfly species in savannah areas of Mato Grosso, Brazil. Therefore, we tested the hypotheses that: (i) higher species richness and (ii) beta diversity would be recorded in more preserved environments, and (iii) species composition would be more homogeneous in disturbed habitats. Preserved sites have wider riparian forest and therefore may provide greater availability and diversity of food resources. We assume that environmental conditions such as temperature and humidity in preserved areas are more favorable for butterflies, in contrast to the observed when the vegetation is removed (higher sunlight input, high temperatures, drought stress and loss of specific microhabitats). The reduction or elimination of resources and/or specific microhabitats would lead to the local extinction of specialist species, favoring the persistence of generalist species.

Material and methods

Our study was carried out in two streams that drain from the Left Bank of the Pindaíba River. This river is a tributary in the right bank of the Middle Mortes River, located at the Southwest region of the state of Mato Grosso, and runs through the municipalities of Barra do Garças, Araguaiana, Cocalinho and Nova Xavantina. The regional climate is Aw according to Köppen, with two well-defined seasons – dry from May to October and rainy from November to April (Peel et al., 2007). The average annual rainfall ranges from 1500 to 1800 mm and the temperature from 18.9 and 33.7 °C (INMET, 2009). The main soil use changes are derived from beef cattle breeding activities and extensive agriculture.

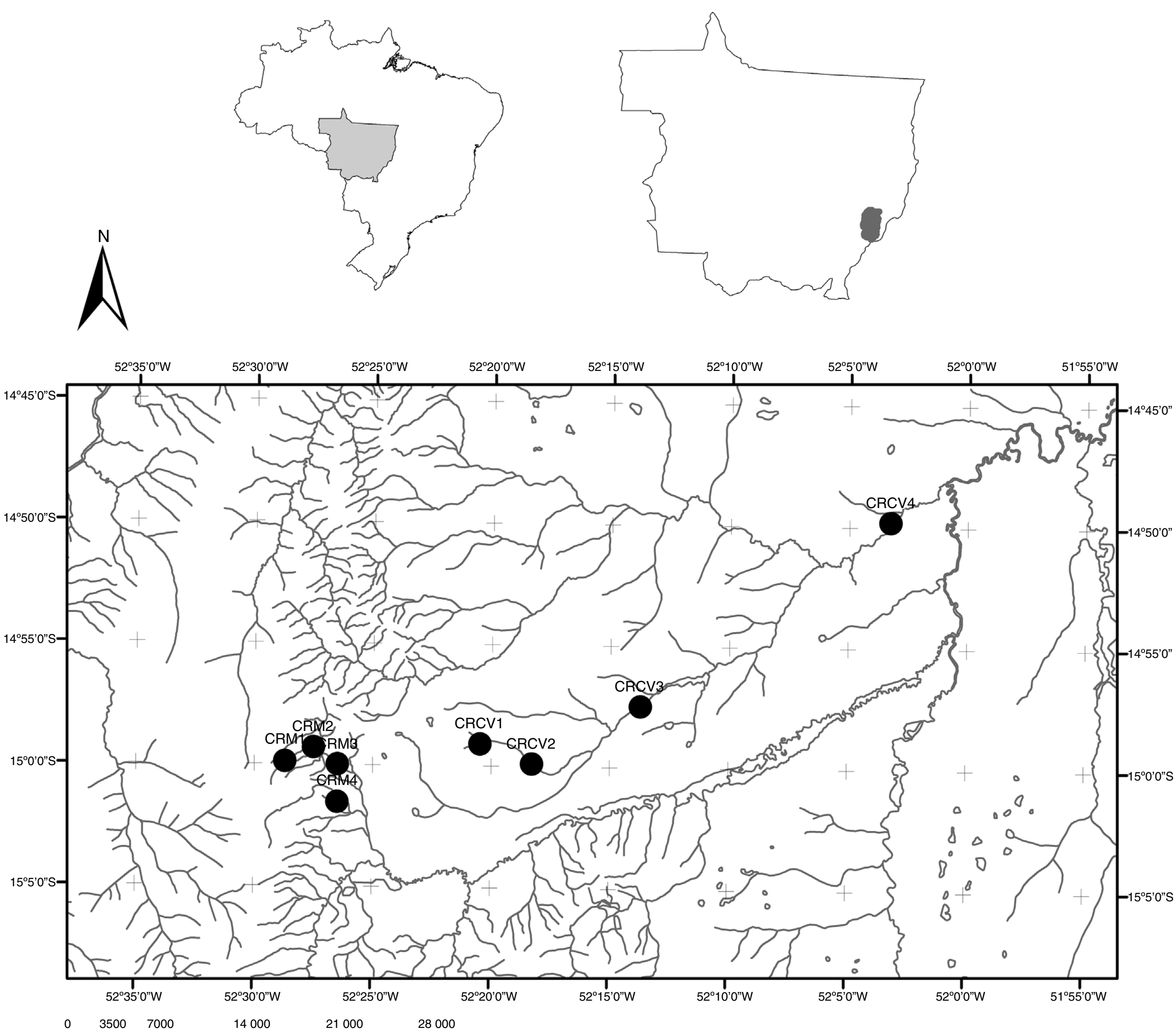

Fig. 1. Butterfly sampling sites at the Pindaíba River Basin, MT – Brazil; (CVS_1, CVS_2, CVS_3, CVS_4 = Caveira stream (1st to 4th order); MS_1, MS_2, MS_3 and MS_4 = Mata Stream (1st to 4th order).

Table 1
Butterfly sampling sites in the Riparian Forest, orders of streams, codes, geographical coordinates and Habitat Integrity Index (HII); Pindaíba River Basin, MT, 2007–2008.

Streams	Orders	Codes	Coordinates	HII
Caveira	1st	CVS_1	S14° 59′ 06.0″ W50° 20′ 29.0″	0.6
	2nd	CVS_2	S14° 59′ 53.4″ W52° 18′ 17.5″	0.5
	3rd	CVS_3	S14° 57′ 28.7″ W52° 13′ 43.9″	0.7
	4th	CVS_4	S14° 49′ 47.7″ W52° 03′ 16.4″	0.6
Mata	1st	MS_1	S14° 29′ 51.7″ W52° 28′ 42.6″	1.0
	2nd	MS_2	S14° 59′ 25.2″ W52° 27′ 57.7″	0.9
	3rd	MS_3	S14° 59′ 59.0″ W52° 26′ 29.0″	0.8
	4th	MS_4	S14° 01′ 37.0″ W52° 26′ 29.0″	0.9

The samples were collected at the shores of the Mata (MS) and Caveira (CVS) Streams, in 1st, 2nd, 3rd and 4th order segments (according to the classification of Strahler, 1957) with different riparian forest conservation levels (eight sampling sites; Fig. 1, Table 1). The Mata Stream still holds most of its riparian forest once it is located in a sharp relief, while the Caveira Stream (located in a mild relief) is considered ideal for agriculture and agricultural mechanization activities, and thus has most of its marginal vegetation changed or removed and dams established for the use of livestock. The samples were collected in two seasons, in each season we collected twice: dry period (August of 2007 and May of 2008) and rainy period (November of 2007 and January of 2008). Samples were collected using dip nets, with the sampling effort of one person collecting per hour, covering fixed 100-m segments along both banks. Each site (order of the streams) was inventoried always between 9 a.m. and 4 p.m. The specimens collected were killed with finger pressure in the chest, stored in properly labeled entomological envelopes and taken to the laboratory to be assembled, identified and preserved dry.

The species were identified using specialized literature (e.g., Brown, 1992; Uehara-Prado et al., 2004; D'Abrera, 1994) and by comparison with specimens stored at the Zoobotanical Collection "James Alexander Ratter" of the Universidade do Estado de Mato Grosso, Nova Xavantina campus. The species identified underwent confirmation of a taxonomist expert in the order, when necessary. The taxonomic classification followed Lamas (2004).

The conservation status of the sites sampled was evaluated using the Habitat Integrity Index (HII) (Nessimian et al., 2008). The HII is a protocol consisting of 12 items that assess environmental conditions, such as: use of the land adjacent to the riparian vegetation, riparian forest width and preservation state, state of the riparian vegetation within a 10 m range, internal structure of the stream (e.g., sediment and retentions). The HII ranges from 0 to 1, where higher values indicate environmental preservation. The HII data obtained were categorized into two conservation categories: disturbed habitats (HII < 0.65 > 0.50) and preserved environments (HII > 0.82). Disturbed habitats were categorized by loss/replacement of original vegetation at least in one of the margins and flow changed by the presence of dams. Preserved habitat were categorized by well-preserved areas, and if had human influence, is beyond the riparian zone.

Statistical analyses

The richness, composition and beta diversity patterns were evaluated by two approaches: considering the conservation metrics (HII values) as categorical variables (conserved/altered) to detect more robust patterns, and as quantitative variables to find subtler patterns of change in the assembly of structure. We used these two metrics intended to reach people with theoretical and scientific interests and those related to public policy and environmental monitoring. The observed species richness is often imprecise due to the difficulty in sampling all species of a given site. Therefore, we used the richness estimated by the nonparametric jackknife estimator, using the software EstimateS Win 7.5.0 (Colwell, 2005). This technique yields more accurate values of species richness of a given community (Krebs, 1999; Walther and Morand, 1998), and provides a confidence interval that enables statistical comparisons between two or more sampled regions. We used the inference by confidence interval to compare richness among sites, where a treatment is only considered different from the other when the confidence interval of a group does not overlap the mean of the other.

Beta diversity was estimated for each site sampled. The relative abundance of each species was determined for each site adding all sampling times. Beta diversity was compared among the streams sampled, separated by station which were considered replicas in hypothesis testing. The approach is based on the dissimilarity measure estimated using the Jaccard quantitative index (Chao et al., 2005). The quantitative index was used, measured the differences in species composition, and is calculated using the total relative abundance of species in each sampled area. According to Chao et al. (2005), the quantitative index is the best beta diversity estimator once it is not dependent on species richness, and is more accurate even for data with small sample sizes. This index considers species abundance, as well as an estimation of the species that were probably not sampled. Higher Jaccard index values correspond to higher turnover rates (more different species composition among sites).

The mean abundance and beta diversity values (among preserved and altered streams) were compared using the Student's *t* test (Zar, 1999). The assumptions of homogeneity and normal distribution were always evaluated before conducting the analyses.

We used a principal coordinates analysis (PCoA) with the Jaccard quantitative index to summarize the structure and community composition data. The PCoA is a method used to explore and visualize similarities or differences in the data using a similarity matrix. The data were log transformed ($\log(x+1)$) to remove the outlier effect and ensure homogeneity of variances. A Permutational Multivariate Analysis of variance (PERMANOVA) (Anderson, 2001) was used to test for differences in species composition between preserved and altered areas. The PERMANOVA uses random permutations (999) based on the Bray–Curtis similarity matrix. We used linear regressions to evaluate the effect of environmental integrity (HII values), on the estimated species richness and on beta diversity (Zar, 1999). A Linear regression between the HII values and the first axis of the PCoA was used to evaluate the relationship between HII and species composition.

Results

Environmental description

The four segments sampled at the riparian forest of the Caveira stream (1st to 4th order – stream width in the segments ranged from 140 to 1720 cm and depth from 13 to 163 cm) had the lowest HII values (0.52–0.65). The segments sampled at the shores of the Mata stream (stream width in the segments ranged from 263 to 486 cm and depth from 33 to 40 cm) yielded higher HII values (0.82–0.96).

Community overview

A total of 245 butterflies were recorded in 32 h of sampling, comprising five families, 63 genera, 84 observed and 133 ± 22.91 estimated species (mean ± confidence interval). The family Nymphalidae was the most well represented, with 29 species sampled, followed by Hesperiidae (25 species), Riodinidae (20), Pieridae (6) and Lycaenidae (4) (Appendix A).

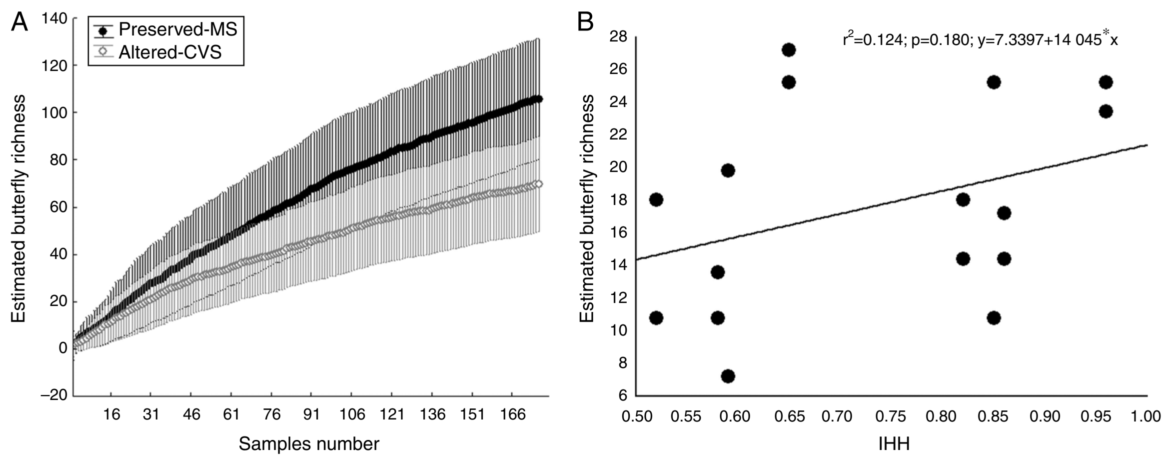

Fig. 2. Relationship between environmental integrity and butterfly species richness in preserved and altered stream at the Pindaíba River basin, MT, Brazil. (A) Species accumulation curve and (B) linear regression.

The most abundant species were *Hermeuptychia hermes* (Fabricius, 1775) (10.9%), *Eurema elathea* (Cramer, 1777) (7.9%), *Pyrgus orcus* (Stoll, 1780) (7.9%), and *Doxocopa agathina* (Cramer, 1777) (6.4%). 45 species were represented by only one individual in the samples (Appendix A).

Environmental integrity and the butterfly community

Species richness (hypothesis I)

We observed a difference between the estimated species richness (conserved = 105.74 ± 25.62; altered = 69.84 ± 20.21; Fig. 2A) when analyzing the categorical metrics of conservation status (altered and conserved). Still, the species accumulation curve of the preserved environments has a steeper slope than the curve of the altered sites, which stabilizes with a smaller number of sites (the preserved sites had on average 35 species more that the altered sites). However, we observed that the estimated species richness was not affected by the conservation level ($r^2 = 0.124$, $p = 0.180$) when we used the quantitative habitat integrity metric (individual HII values) to seek subtler structure patterns (Fig. 2B).

Beta diversity (hypothesis II)

The beta diversity values differed between the preserved and altered sites ($t = 2.771$, gl = 4, $p = 0.015$). The beta diversity value of preserved sites was 0.07 higher than that of altered sites, showing a higher turnover among preserved sites than among altered sites. There was also a positive effect of the integrity index on beta diversity ($r^2 = 0.319$, $p = 0.022$) (Fig. 3).

Species composition (hypothesis III)

We recorded differences between the preserved and altered treatments when ordering the sampled sites with respect to the conservation category (Fig. 4A), which was also confirmed by the PERMANOVA ($F = 2.306$; $R^2 = 0.141$; $p < 0.001$). The altered sites were more clustered in the ordination space (graph), indicating a more homogeneous species composition and a large similarity among altered sites, corroborating our third hypothesis. On the other hand, the preserved sites had a wider turnover, which indicates more pronounced fauna heterogeneity when compared with altered sites. A total of 23 of the 84 species sampled occurred only in altered environments, 42 were exclusive to preserved sites and 19 occurred in both stream categories. The habitat integrity measured by the HII was also positively related with the first axis of the PCoA, which summarizes the variation of species composition among the sampled sites ($r^2 = 0.521$, $p = 0.002$) (Fig. 4B).

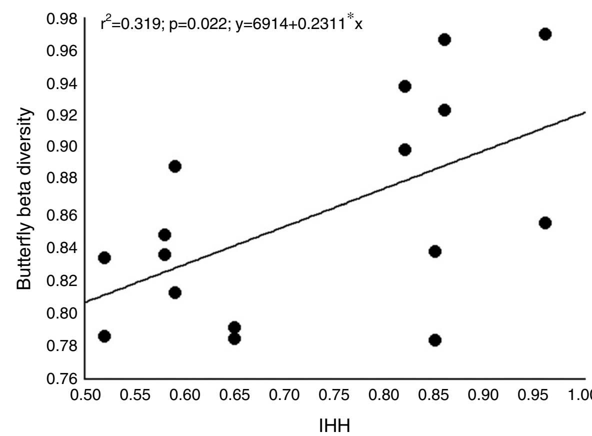

Fig. 3. Relationship between beta diversity of butterfly species in preserved and altered streams and the Habitat Integrity Index (HII), of the Pindaíba River basin, MT – Brazil.

Discussion

The Cerrado biome has heterogeneous landscapes with several plant formations and is a habitat with the conditions and resources required for the survival of a large variety of butterflies that occur in this environment (Emery et al., 2006). The highest species richness of the Cerrado biome occurs in humid areas, such as riparian forests, once such areas provide resources that are rarer or absent in the more open adjacent systems (e.g., less humid and rich soil) (Brown, 2000). The species of the subfamily Ithomiinae, for example, need environments with nutrient rich soils and abundant water to develop, and are faithful to their microhabitas due to its specific requirements (Brown, 1992; Brown and Freitas, 2000, 2002). Therefore, the environmental specificities of the riparian forest area (meeting specific biological requirements) could explain the diversity of species recorded in our study. In a recent study, Queiroz-Santos et al. (2016) produced the historical surveys of species of butterflies recorded for Mato Grosso, and mentioned a richness between 51 and 100 species in the region, indicating that our data is representative for the regional fauna. The state of Mato Grosso is potentially highly biodiverse because three of the main Brazilian biomes are present within its borders: Amazon tropical rainforest, Cerrado and Pantanal. However, little information is available on the magnitude of their biodiversity. The greater species richness recorded in preserved areas may also be explained by the high diversity and availability of microhabitats

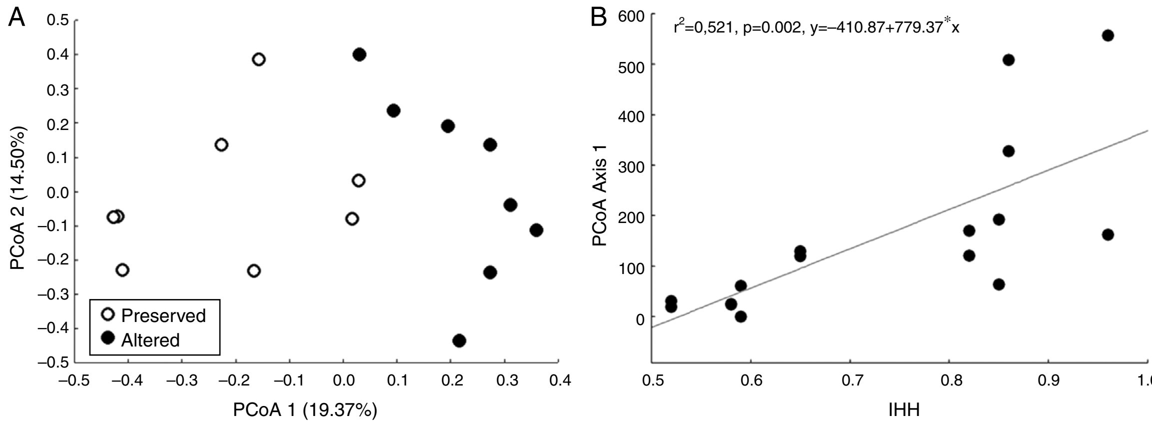

Fig. 4. (A) Ordination of the sampled sites in regards to butterfly composition and abundance and (B) relationship between beta diversity of butterflies and the Habitat Integrity Index (HII), Pindaíba River basin, MT – Brazil.

in the riparian forest. Natural riparian zones are some of the most diverse, dynamic, and complex biophysical habitats on the terrestrial portion of the planet (Naiman et al., 1993), riparian forests act as refuges in adjacent areas and, in some cases, as corridors for migration and dispersal (Naiman and Décamps, 1997). Therefore, some vegetation types would be more important than others for conservational purposes. The authors grouped the localities for phytophysiognomy, and verified that riparian and Cerradão fauna were similar and form a distinct group from those with open crown (savannas). They suggest that the type of vegetation is the one of the most important factors determining assemblies of Arctiidae in Cerrado. On the other hand, Ferro and Romanovski (2012) compared forest border and Grassland areas in Rio Grande do Sul, and found Grassland most rich, probably because of nectar supply available in herbaceous of the area.

In this study, it is likely that species specific to each microhabitat exist. Such species would have a higher specialization in more heterogeneous and preserved riparian forest environments to occupy unique suitable niches preferred for population maintenance.

The lack of relationship between HII, and composition and richness when using quantitative variables may be a consequence of the high flight capacity and migration of some butterfly species (Marini et al., 2009; Shahabuddin and Terborgh, 1999; Veddeler et al., 2005). An example is the species *Danaus plexippus* (Linnaeus, 1758), which can fly over about 3000 km, migrating in search of areas with milder environmental conditions changing period and flyway to avoid unfavorable climatic events of that particular year (Miller et al., 2012). A high flight capacity may enable the organisms originally from altered sites to enter the riparian forest seeking other resources, and may allow the species to seek environmental characteristics more similar to their niche. Therefore, the realized niche would be wider for species with higher flight/migration capacity. However, Komonen et al. (2004) proved that more specialist butterfly species have lower mobility and rarely leave the habitat location where the food supply for the larval stage is located whereas less specialist species may have higher mobility. Still, when the increased land use results in a homogeneous landscape, the altered area becomes the habitat of species with higher mobility (generalist species) (Dormann et al., 2007). Therefore, the specialist species would be restricted to vegetation patches and strong habitat changes may cause local extinction, once fragmentation and habitat modification would create dispersal barriers.

We may consider this divergence between the results obtained from categorical and quantitative changes to be an indication that species richness may not be a good metric to assess habitat quality, especially given that beta diversity and species composition were influenced by the same changes in the riparian forest that were considered in testing species richness. Species richness may not be a good parameter to study the effects of environmental changes on butterflies, once it considers only the number of species regardless of the identity or ecological role of the species entering or leaving the community. The species richness of altered environments may be similar to that of pristine environments, once richness may remain high due to the entrance of generalist instead of specialist species with low conservation value. Note that specialist species may occur in several types of environments. On the other hand, beta diversity and species composition have proved to be effective to capturing subtle community changes.

The hypothesis that beta diversity of butterflies would be higher in preserved environments was corroborated. The preserved areas of our study have higher environmental heterogeneity than altered, and possibly allow the coexistence of species with different requirements, enabling greater habitat sharing which in turn leads to greater beta diversity. These environments, when altered by native vegetation removal, received grazing or extensive monoculture, getting simpler and more homogeneous. In such context, few species manage to survive, usually those with fewer environmental demands and wider niches, leading to a decrease in regional beta diversity. However, this pattern is not always observed. The high migration capacity of butterflies tends to homogenize the spatial scale effect and limit the composition variation patterns (Marini et al., 2009). The restrictions of specialist species to the humid areas of the Cerrado biome may have effectively contributed to such result.

Beta diversity was positively related with HII, showing that preserved sites have a higher variation in species composition than the altered sites, which in fact, was expected. Our results support the study carried out by Ekroos et al. (2010), who state that agricultural intensification reduces the beta diversity of butterflies, regardless of its location in the river basin. We also believe that the number of rare species (42 exclusive to preserved sites and 19 to altered sites) contributed effectively to the observed beta diversity variation, once the landscape-scale beta diversity measures are more affected by rare species than the alpha diversity. The specialist species are precisely those that most suffer the effects of landscape changes (Marini et al., 2009).

Likewise, the hypothesis that species composition would be more homogeneous in altered environments was corroborated. This result is probably a consequence of the greater light input in altered sites, which causes greater water stress and loss of specific microhabitats, with the elimination of specialist species and entrance of generalist species. In addition, the higher light input in altered sites reduces humidity and contributes to higher temperature oscillation, which can prevent the occurrence of stenotopic butterflies.

The restoration of butterfly communities in an area that underwent natural a disturbance is rapid and complex. However, most butterfly species disappear altogether in profoundly changed, anthropized or polluted environments, leaving few resistant, adaptable or colonizing species able to reach high population densities (Brown and Freitas, 1999). In addition, both alpha and beta diversity reduce more quickly when natural vegetation is smaller than 40%, increasing the negative effects on the butterfly diversity (Ekroos et al., 2010).

The little vegetation present in disturbed habitats may have provided a different microhabitat for the species. However, these small vegetation patches may not accommodate the local fauna for a long time, due to increased susceptibility to stochastic processes. Therefore, the alteration of riparian vegetation directly affects the diversity and composition of butterfly species of the Cerrado. The conservation of the riparian forest is therefore essential for the conservation of the aquatic fauna as well as for the terrestrial fauna, as we demonstrated. Thus, it is extremely important that the riparian vegetation not be altered by soil use activities. On the other hand, deforested areas should be reforested seeking the restoration of ecosystem services, enabling gene flow among sites.

Conflicts of interest

The authors declare no conflicts of interest.

Acknowledgements

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Appendix A.

List of the Lepidoptera (butterflies) species found at the Caveira (CVS) and Mata (MS) streams, MT–Brazil, August/2007–May/2008 [1–4 = order of the streams].

Species	Family	CVS1st	CVS2nd	CVS3rd	CVS4th	MS1st	MS2nd	MS3rd	MS4th	Total
Adelpha iphiclus (Linnaeus, 1758)	Nymphalidae	–	1	–	–	1	–	–	–	2
Adelpha thesprotia Felder & Felder, 1867	Nymphalidae	–	–	–	–	–	–	–	1	1
Aguna albistria albistria (Plötz, 1880)	Hesperiidae	–	–	–	–	–	1	–	–	1
Aguna sp.	Hesperiidae	–	–	–	–	1	–	–	–	1
Anartia jatrophae jatrophae (Linnaeus, 1763)	Nymphalidae	–	5	–	1	–	–	–	–	6
Antigonus erosus (Hübner, 1812)	Hesperiidae	–	–	–	–	1	–	1	–	2
Aphrissa statira (Cramer, 1777)	Pieridae	–	1	1	1	3	–	–	–	6
Aricoris campestris (H. Bates, 1868)	Riodinidae	4	2	1	1	–	–	–	–	8
Aricoris sp.	Riodinidae	–	–	1	–	–	–	–	–	1
Autochton neis (Geyer, 1832)	Hesperiidae	–	–	–	–	–	1	–	–	1
Brevioleria aelia plisthenes (R.F. d’Almeida, 1958)	Nymphalidae	–	–	–	–	1	–	–	–	1
Brevioleria seba emyra (Haensch, 1905)	Nymphalidae	–	–	–	–	4	–	–	–	4
Caligo illioneus Butler, 1870	Nymphalidae	–	–	–	–	1	–	–	–	1
Caligo teucer (Linnaeus, 1758)	Nymphalidae	–	–	–	–	1	–	–	–	1
Calopsila lucianus (Fabricius, 1793)	Riodinidae	–	–	–	–	–	–	–	1	1
Calycopis bellera (Hewitson, 1877)	Lycaenidae	1	–	–	–	–	–	–	–	1
Calycopis sp.	Lycaenidae	–	–	1	–	–	–	–	–	1
Caria plutargus (H. Bates, 1868)	Riodinidae	–	–	–	–	1	–	–	–	1
Chamaelimnas tircis C. Felder & R. Felder, 1865	Riodinidae	–	–	1	–	–	–	–	–	1
Chioides catilus catilus (Cramer, 1779)	Hesperiidae	–	–	–	–	–	–	1	–	1
Chorinea amazon amazon (Saunders, 1859)	Riodinidae	–	–	–	–	–	1	1	–	2
Coeruleuptychia brixus (Godart, [1824])	Nymphalidae	–	–	–	–	–	–	1	–	1
Detritivora cuiaba (Harvey & Hall, 2002)	Riodinidae	1	–	2	–	–	2	–	–	5
Diaethria clymena meridionalis (Bates, 1864)	Nymphalidae	–	–	–	–	–	–	1	1	2
Doxocopa agathina Cramer, 1777	Nymphalidae	–	–	–	–	1	14	–	2	17
Eueides vibilia unifasciatus A. Butler, 1873	Nymphalidae	–	–	–	–	1	–	–	–	1
Eunica sydonia Godart, 1824	Nymphalidae	1	–	–	–	–	–	–	–	1
Eurema albula (Cramer, 1775)	Pieridae	–	–	–	–	–	–	1	1	2
Eurema deva (Doubleday, 1847)	Pieridae	–	–	–	–	–	1	–	–	1
Eurema elathea (Cramer, 1777)	Pieridae	5	4	4	5	1	–	1	1	21
Eurybia elvina emidiata Stichel, 1915	Nymphalidae	–	–	–	–	1	–	–	–	1
Eurybia sp.	Nymphalidae	–	–	–	–	1	–	–	–	1
Evenus satyroides (Hewitson, 1865)	Nymphalidae	–	–	–	–	–	–	1	–	1
Exoplisia cadmeis (Hewitson, 1866)	Riodinidae	–	–	–	1	–	–	–	–	1
Gorgythion begga begga (Prittwitz, 1868)	Hesperiidae	1	–	–	–	1	–	–	–	2
Gorgythion sp.	Hesperiidae	–	–	–	–	–	3	–	–	3
Hamadryas arete (Doubleday, 1847)	Nymphalidae	–	–	–	–	–	–	1	–	1
Helias phaloenoides palpalis (Latreille, [1824])	Nymphalidae	–	–	–	–	1	–	–	–	1
Heliconius erato phyllis (Fabricius, 1775)	Nymphalidae	–	–	1	–	2	1	1	2	7
Heliconius melpomene nanna Stichel, 1899	Nymphalidae	–	–	–	–	–	–	–	1	1
Heliconius sara thamar (Rübner [1806])	Nymphalidae	–	–	–	–	1	1	–	–	2
Heliopetes arsalte ersalte (Linnaeus, 1758)	Hesperiidae	–	1	2	–	–	–	–	–	3

Species	Family	CVS1st	CVS2nd	CVS3rd	CVS4th	MS1st	MS2nd	MS3rd	MS4th	Total
Hemiargus hanno hanno (Stoll, 1790)	Lycaenidae	–	2	2	6	–	–	2	2	14
Hermeuptychia hermes (Fabricius, 1775)	Nymphalidae	8	1	5	–	–	–	–	8	22
Hesperiidae sp1.	Hesperiidae	–	–	–	–	–	1	–	–	1
Hesperiidae sp2.	Hesperiidae	1	–	–	–	–	–	–	–	1
Hesperiidae sp3.	Hesperiidae	–	–	–	–	1	–	–	–	1
Hyphilaria parthenis (Westwood, 1851)	Riodinidae	–	–	–	1	–	–	1	–	2
Hypoleria lavinia (Hewitson, [1855])	Nymphalidae	–	–	–	–	1	–	–	–	1
Junonia evarete (Cramer, 1779)	Nymphalidae	1	–	–	1	–	–	–	–	2
Lento apta Evans, 1955	Hesperiidae	–	–	–	–	–	1	–	–	1
Libytheana carinenta (Cramer, 1777)	Nymphalidae	–	–	–	–	–	2	–	–	2
Mechanitis polymnia casabranca	Nymphalidae	–	–	–	–	–	1	–	–	1
Melanis smithiae smithiae (Westwood, 1851)	Riodinidae	–	–	1	–	–	–	–	1	2
Mesosemia synnephis Stichel, 1924	Riodinidae	1	–	–	–	–	–	–	–	1
Morpho helenor Cramer, 1776	Nymphalidae	–	–	–	–	2	–	1	–	3
Morpho menelaus (Linnaeus, 1758)	Nymphalidae	–	–	–	–	2	–	–	–	2
Nymphidium caricae caricae (Linnaeus, 1758)	Riodinidae	–	–	2	–	–	–	–	1	3
Nymphidium lisimon epiplatea A. Butler, 1867	Riodinidae	–	–	1	–	–	–	–	1	2
Nymphidium nivea Talbot, 1928	Riodinidae	–	–	3	2	–	–	–	–	5
Ouleos fredericus candangus Mielke, 1968	Hesperiidae	–	1	–	–	1	–	–	–	2
Panoquina sp.	Hesperiidae	–	–	–	–	–	–	1	–	1
Parcella amarynthina (C. Felder & R. Felder, 1865)	Riodinidae	–	–	1	–	–	–	–	–	1
Pareuptychia hesionides Forster, 1964	Nymphalidae	–	–	1	–	–	1	–	–	2
Pellicia sp.	Hesperiidae	–	1	–	–	–	–	–	–	1
Phoebis sennae sennae (Linnaeus, 1758)	Pieridae	–	–	1	–	–	–	–	–	1
Polites vibex catilina (Plötz, 1886)	Hesperiidae	–	–	1	–	–	–	–	–	1
Pompeius pompeius (Latreille, [1824])	Hesperiidae	2	–	–	–	–	–	–	1	3
Pyrgus orcus (Stoll, 1780)	Hesperiidae	4	3	4	7	1	–	–	2	21
Pyrisitia nise Cramer, [1775]	Pieridae	–	–	2	1	–	1	–	1	5
Pythonides herennius herennius Geyer, [1838]	Hesperiidae	–	–	–	–	–	–	1	–	1
Quadrus cerialis (Stol, 1782	Hesperiidae	–	1	–	–	–	–	–	–	1
Stalachtis phlegia phlegetontia (Perty, 1833).)	Lycaenidae	–	–	1	1	–	–	2	–	4
Synargis axenus (Hewitson, 1876)	Riodinidae	–	–	–	–	–	1	–	–	1
Synargis calyce (C. Felder & R. Felder, 1862)	Riodinidae	–	–	1	–	–	–	–	–	1
Taygetis echo (Cramer, 1775	Nymphalidae	–	–	1	–	–	–	–	–	1
Tegosa claudina (Eschscholtz, 1821)	Nymphalidae	–	–	–	–	–	–	2	1	3
Thoon sp.	Hesperiidae	–	–	–	–	–	–	–	1	1
Tithorea harmonia Cramer, 1777	Nymphalidae	–	–	–	–	1	1	–	–	2
Urbanus dorantes dorantes (Stoll, 1790)	Hesperiidae	–	–	1	–	–	–	–	–	1
Urbanus simplicius (Stoll, 1790)	Hesperiidae	1	2	–	–	–	–	–	–	3
Viola violella (Mobille, 1897)	Hesperiidae	–	–	–	1	–	–	–	–	1
Yphthimoides affins (Butler, 1867)	Nymphalidae	–	–	1	–	1	–	–	–	2
Zaretis itys (Cramer, [1777])	Nymphalidae	–	–	–	–	–	–	1	–	1
Total		**31**	**25**	**43**	**29**	**34**	**34**	**21**	**29**	**246**

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