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Beta diversity of plant–herbivore interactions is unaffected by urbanization levels in Brazilian Cerrado

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Abstract

Plant–herbivore interactions are pivotal in shaping terrestrial ecosystems, influencing plant populations and insect diversity; however, little is known about how anthropogenic impacts affect the beta diversity of these interactions. In our study, we investigated plant–herbivore networks across an urbanization gradient in Brazilian Cerrado. We tested two hypotheses: (1) urbanization decreases interaction dissimilarity, and (2) herbivorous insects show greater dissimilarity than plants. To test these hypotheses, we conducted data collection across 16 sites, representing different urbanization levels—urban, rural, and wild. We sampled plant–herbivore interactions for 310 insect herbivore species and 97 host plant species. Our analysis revealed that beta diversity of interactions was consistently high across all environments studied. However, we did not find any significant differences in total interaction dissimilarity among the different levels of urbanization. We found that the primary driver of dissimilarity was species composition turnover, with herbivorous insects contributing more to dissimilarity. Our findings challenge the conventional wisdom that urbanization significantly alters plant–herbivore interactions. Instead, we observed consistent interaction dissimilarity, highlighting the resilience of ecological networks in the face of anthropogenic impacts. Our results underscore the complexity of these interactions and emphasize that plant–herbivore interactions can exhibit a high degree of dissimilarity even in urban environments.

Keywords Ecological networks · Interaction turnover · Neotropical savannas · Plant–animal interactions

Introduction

Plant–herbivore interactions are fundamental ecological processes that shape the structure and dynamics of terrestrial ecosystems (de Vries et al. 2017). The great diversity of interactions between herbivores and plants can generate complex ecological networks, wherein species are depicted as nodes and their interactions as links connecting them (Poisot et al. 2015). These interactions involve a wide array of animals, mainly insects, and play a crucial role in regulating plant populations, nutrient cycling, and energy flow within ecosystems (Yang and Gratton 2014; Kozlov and

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Zvereva 2017). Insect herbivores exert selective pressure on plants, influencing plant community composition, diversity, and the evolution of plant defense strategies (Poelman 2015). Herbivorous insects also depend intrinsically on their host plants, which can influence their survival and evolution (Lewinsohn et al. 2005). Plants provide not only shelter but also essential sustenance for these insects, and any alteration in plant composition can have a significant impact on their ecology (Underwood et al. 2014). Variations in host plant availability can affect both the quantity and quality of food resources for herbivorous insects (Bröcher et al. 2023). Given the high diversity of herbivorous insects and host plants, a high turnover of plant and herbivorous species among local communities is also expected, resulting in a high beta diversity in plant–herbivore assemblages (Lewinsohn et al. 2022; Belchior et al. 2023).

Beta diversity quantifies the dissimilarity of species composition between different sites (Anderson et al. 2006). In the context of plant–herbivore assemblages, beta diversity reflects the turnover of plant and herbivore species among habitats (Novotny 2009). High beta diversity indicates significant dissimilarity in species composition between sites, suggesting pronounced turnover of species and potentially different ecological processes occurring in each location. When comparing plant and insect herbivore beta diversity, we can observe intriguing differences in the levels of dissimilarity (Kemp et al. 2017; Martins et al. 2019). This variability in dissimilarity arises, in part, from the inherent asymmetry of dependence within herbivory interactions. Specifically, herbivorous insects, heavily rely on plants for sustenance, while plants are not contingent on the presence of consumers for their survival (Price 2002). Given this intrinsic imbalance, herbivorous insects frequently manifest more pronounced responses to ecological shifts and community alterations than host plants (Tscharntke and Hawkins 2002). Consequently, modifications in the species composition of plant communities can initiate a cascade of consequences throughout the ecological network, potentially intensifying beta diversity at higher trophic levels (Martins et al. 2019). In light of these intricate ecological dynamics, it becomes reasonable to anticipate that within plant–herbivore networks, the dissimilarity of interactions will exhibit greater disparities among herbivorous insects than among host plants. This expectation aligns with the heightened sensitivity of herbivores to ecological perturbations and their profound dependence on the nuanced composition of plant species for sustenance. Although studies evaluating the taxonomic beta diversity of insect herbivores and plants are not rare, investigations into how the beta diversity of interactions varies in space and time remain scarce (Tylianakis and Morris 2017; Belchior et al. 2023).

The beta diversity of interactions often exceeds the beta diversity of species because, as more networks are examined across different locations and times, new interactions between previously known species are continually being uncovered (Lewinsohn et al. 2022). Beta diversity of interactions can be understood as the turnover of interactions between ecological networks (Poisot et al. 2012). This phenomenon is proposed to arise from two interconnected components: species turnover and interaction rewiring (Tylianakis and Morris 2017). Species turnover involves changes in interactions within a community due to variations in species composition across different environmental conditions, leading to alterations in network structure (CaraDonna et al. 2017). Interaction rewiring, on the other hand, stems from the reconfiguration of interactions between the same species under differing circumstances, such as seasonal changes or shifts between habitats (Poisot et al. 2012; CaraDonna et al. 2017). Understanding these components is crucial for unraveling the dynamics of ecological networks and their responses to environmental changes, underscoring the need for further research in this field (Tylianakis and Morris 2017). In this context, a question that has been little explored so far is how anthropogenic impacts affect the beta diversity of plant–herbivore interaction networks.

Human disturbances, such as habitat fragmentation and human occupation, can greatly impact plant communities, leading to changes in species composition and reduced beta diversity (Araújo et al. 2022). The human-induced habitat modification creates fragments of natural vegetation subjected to different types of landscapes, such as rural and urban environments (Hutchings et al. 2022; Salamanca-Fonseca et al. 2024). In fragments of natural vegetation located in rural areas, high diversities of plants (Freitas et al. 2020) and herbivorous insects (Araújo et al. 2024) are generally found, with these environments often being similar in terms of diversity to wild environments (Araújo et al. 2024; Salamanca-Fonseca et al. 2024). On the other hand, urban environments often support a limited number of plant species, characterized by homogenized and simplified plant communities (Dylewski et al. 2023). For instance, urban fragments may be dominated by a few species of plants usually tolerant to urban matrix, whereas fragments in natural landscapes encompass a broader spectrum of plant species (Freitas et al. 2020). The loss of plant species can affect the available resources for herbivores and influence their species composition and abundance. Several studies suggest that plant–insect interactions could undergo significant alterations due to the incapacity of many plant and insect species to tolerate the biotic and abiotic changes associated with urban environments (Concepción et al. 2015; Miles et al. 2019; Araújo et al. 2024). In urban areas, common native plant species may be replaced by non-native plants, leading to changes in the abundance and distribution of herbivorous insects (Martin and Wilsey 2012; Araújo et al. 2015). The adverse impacts of urbanization may be particularly pronounced for insects with specialized diets, as their host plants could also suffer significant negative consequences, such as decreased abundance and changes in plant defenses (Moreira et al. 2019; Miles et al. 2022). Consequently, with few species interacting and the prevalence of those less specialized, the beta diversity of plant–herbivore interactions in urbanized areas may be lower compared to more diverse and natural habitats.

In this study, our objective is to investigate the beta diversity of interactions within plant–herbivore networks across an urbanization gradient in Brazilian Cerrado. Recent estimates suggest that approximately 40–55% of the Cerrado vegetation has been altered due to human-induced habitat fragmentation, mainly for expansion of agricultural areas and urban centers (Lahsen et al. 2016; Colli et al. 2020). In this context, we tested the hypothesis that the level of urbanization (wild, rural, and urban areas) affects the dissimilarity of plant–herbivore interactions. Due to the role of human activities as filters for both plant and insect species and their interactions in anthropogenic environments, we expect a higher beta diversity of interactions in wild and rural environments compared to urban ones. In addition, we also tested whether plants and herbivorous insects exhibit different dissimilarity levels. Due to the accumulation of biotic filters across different trophic levels, we expect a higher beta diversity for herbivorous insects than for host plants, regardless of the level of urbanization.

Material and methods

Study areas

The study was conducted within the Neotropical savanna areas (cerrado sensu stricto) of northern Minas Gerais State, Brazil (Fig. 1). Neotropical savanna vegetation predominantly consists of sclerophyllous plants that thrive in nutrient-poor soils, where both water and mineral nutrient availability are limited (Ribeiro and Walter 2008). The regional climate falls under the category of dry tropical (Aw in the Köppen system), characterized by distinct rainy seasons, an average temperature of 24.2 °C, and an annual precipitation of 1000 mm (Alvares et al. 2013). Our study encompassed the examination of 16 savanna remnants, located in regions with different land use types:

Urban areas: These are located within the urban region of Montes Claros city (16°43′ S, 43°52′ W), Minas Gerais, Brazil. It covers an area of 3,589.811 square kilometers and has a population of approximately 414,240 people. The population density is approximately 115.4 inhabitants per square kilometer. Montes Claros has experienced significant urbanization, with 95% of its population residing in urban areas. Urban savanna remnants are susceptible to various human-induced activities like pollution by sewage, garbage dump, and fire for debris burning, which can lead to disturbances in the vegetation. In this study, these areas are categorized as having high-intensity land use. We conducted sampling in four urban environments located at least 2 km apart from each other.

Rural areas: These areas are located more than 10 km away from urban zone (16°23′ S, 44°07′ W) and experience low human interference, with livestock grazing, wood harvesting, and occasional instances of fire. As a result, these areas are classified as having low land use. Five areas in this environment were included in our sampling (located at least 2 km apart from each other).

Wild areas: These are remnants of savanna found within the conservation units of Serra do Cabral State Park (17°43′ S, 44°20′ W), situated in the municipalities of Buenópolis and Joaquim Felício, and Veredas do Peruaçu State Park (14°58′ S, 44°39′ W) in the municipality of Bonito de Minas, all located in northern Minas Gerais. These areas are considered completely preserved. We defined seven specific areas within these conservation units (three in Veredas do Peruaçu State Park and four in Serra do Cabral State Park) as shown in Fig. 1. The minimum distance between these areas ranged from 1 to 5 km.

Data collection

Plant–herbivore interactions were sampled in 80 randomly selected plots, each measuring 100 square meters (10 × 10 m), distributed across 16 areas (being five plots per area). These plots were situated at a minimum distance of 20 m from one another and at least 30 m away from the edge of the forest fragments. Within each of these plots, we sampled all tree and shrub individuals with a circumference at breast height (CBH), i.e., 1.30 m above the ground, equal to or greater than 15 cm. Plant species identification was primarily conducted on-site. However, for any unidentified samples, we collected specimens for subsequent identification, employing conventional herborization techniques. Taxonomic classification into families adhered to the Angiosperm Phylogeny Group IV (APG IV 2016) guidelines, and botanical material identification was accomplished through consultation of specific literature and, when necessary, by seeking expert assistance. Species nomenclature and author abbreviations were consistent with the Flora Brasil 2020 online databases (http://floradobrasil.jbrj.gov.br) and The Plant List (http://www.theplantlist.org/).

We conducted collections of herbivorous insects during both dry (July to September) and rainy (October to December) seasons of 2018 and 2019. Collecting during both seasons was deemed necessary for a comprehensive characterization of herbivore assemblages, as seasonality can influence the presence of certain species (Silva et al.

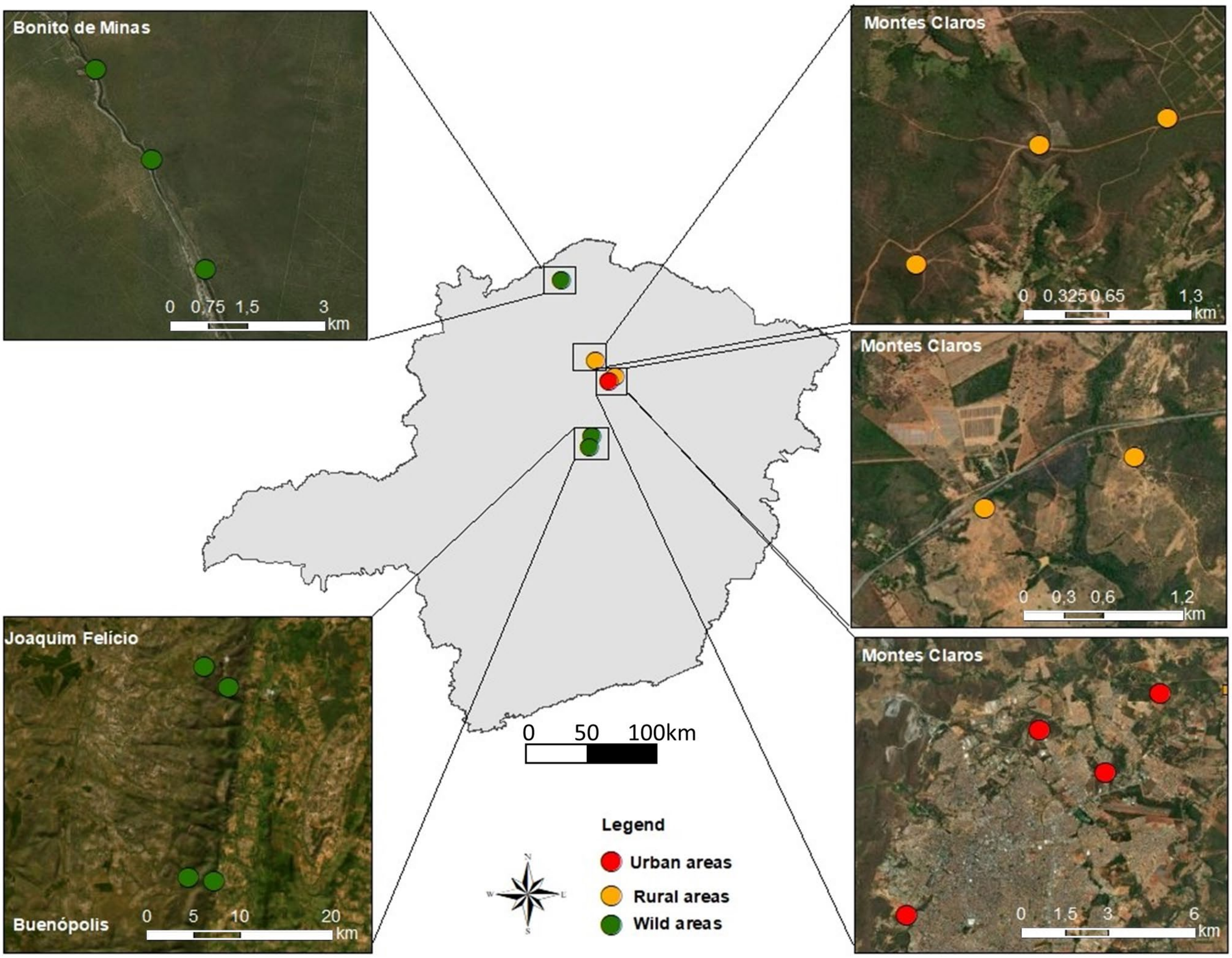


Fig. 1 Location of neotropical savanna areas sampled in the northern region of Minas Gerais, Brazil. In green circles, there are the wild areas in the Serra do Cabral State Park (municipalities of Buenópolis and Joaquim Felício) and Veredas do Peruaçu State Park (municipality of Bonito de Minas), in orange circles, the rural areas of the municipality of Montes Claros, and in red circles, there are the urban areas located in the region within the city of Montes Claros

2011). Within each plot, we selected three branches from each marked plant and subjected them to the tapping method with an entomological umbrella (Oliveira et al. 2020). This method consists of performing 10 strikes with a 1 m wooden stick on each of the branches, causing phytophagous insects to fall from the plant into the entomological umbrella. Specimens collected were preserved in 70% alcohol and placed in containers that were appropriately labeled with information about the area, plot, and the plant from which they were collected. Subsequently, these specimens were transported to the laboratory for sorting and insect identification.

The insects were categorized into morphospecies based on their external morphological characteristics and were identified to the most precise taxonomic level possible. In addition, we separated them based on their developmental stage, distinguishing between adults and immatures. Adult insects were identified using the taxonomic keys of Triplehorn and Johnson (2011), whereas immatures were identified following the guidelines provided by Costa et al. (2006). Adult insects and nymphs were stored individually in entomological boxes with corresponding cataloging, while larval forms were cataloged and placed in Eppendorf tubes containing 70% alcohol for preservation. Our selection of insect taxa for constructing interaction networks focused solely on those known to be phytophagous, following information available in the literature (e.g., Carrano-Moreira 2015).

Data processing

We built matrices A, where Aij = number of times in which an herbivore insect species j was found in association with a plant species i for each of the 16 sampling sites. We calculated the beta diversity of interactions among networks of all sampled sites within each urbanization level (*i.e.,* urban, rural, and wild). Beta diversity of interactions is part of biodiversity and allows us to explore differences in composition and intensity of interactions among sites (Luna et al. 2020). It is calculated following the formula $\beta WN = \beta ST + \beta OS$, where βWN is the total dissimilarity between two networks, βST is the dissimilarity (component) explained by difference in species community composition and βOS is the dissimilarity (component) explained by rewiring among shared species (Poisot et al. 2012). We reached a total of 15 pairwise interaction dissimilarities values using the *bipartite* package's function *betalinkr_multi*. We choose "commondenom" partitioning method to ensure additivity of interactions beta diversity components (*i.e.,* βST and βOS) and used Sorensen dissimilarity index (Novotny 2009; Legendre 2014). We also calculated the proportional contribution of each components dividing βST and βOS by βWN.

To calculate taxonomic beta diversity of both, herbivore insects and plants among sites within each urbanization level, we built matrices A, where $Aij = 1$ when an herbivore or a plant j occurred in a site i and 0 when no occurrence was registered. Taxonomic beta diversity is variation of the species composition of assemblages and can be partitioned into spatial turnover and nestedness components following the formula $\beta SOR = \beta SIM + \beta SNE$. In the formula, βSOR is the total dissimilarity, βSIM is the dissimilarity explained by spatial turnover, and βSNE is the dissimilarity explained by nestedness (species loss) (Baselga 2010). Thus, we calculate the values of taxonomic beta diversity using the *betapart* package's function *beta.pair* configured for Sorensen/Simpson pairwise dissimilarity index (Baselga and Orme 2012).

Statistical analysis

To test the hypothesis that urbanization level affects interaction dissimilarity, we generate a generalized linear model (GLM), where urbanization level (urban, rural, and wild) was the independent variable and βWN was the dependent variable. We explored the contribution of each trophic level (plants and insects) beta diversity for interaction dissimilarities due to changes in species composition. To test if the mean taxonomic beta diversity of both, herbivores insects and plants change among urbanization levels and, which of them are more dissimilar among sites within each urbanization level, we fitted a complete GLM, where trophic level interacting with urbanization level was set as independent variables and βSOR as dependent variable. We simplified the model and selected the final model as follows: βSOR ~ trophic level (insects and plants). We adjusted all models' residuals distribution for Gaussian family and evaluated the dispersion parameters using graphical analysis through Q–Q plot and dividing the models' residual deviance by the degrees of freedom. All analysis was conducted in R Software (R Development Core Team 2023).

Results

The plant–herbivore interaction networks were characterized by 97 plant species, 310 insect herbivore species, and 500 distinct interactions. The Fabaceae family emerged as the most prominent, boasting 23 species, followed by Vochysiaceae with ten species and Myrtaceae with six species. The recorded herbivorous insects belonged to six orders: Coleoptera, Hemiptera, Lepidoptera, Orthoptera, Phasmatodea, and Thysanoptera. The richest insect orders were Hemiptera with 137 species and Coleoptera with 102 species. The number of plant species in networks varied from 11 to 38 species, with an average of 194 (±7.5), while the number of insect species varied from eight to 59 species (27.6 ± 16.1).

In total, 39 plant species were recorded in wild environments, with an average of 14.8 (±5.0) species per area. In rural environments, 35 plant species were recorded (11.0 ± 1.6 per area), while in urban environments, 32 plant species were recorded (10.0 ± 5.2 per area). For herbivorous insects, 152 species were recorded in wild environments (45.0 ± 20.1 per area), 88 in rural environments (19.8 ± 4.2), and 85 in urban environments (25.0 ± 16.4).

The beta diversity of interactions was elevated in all studied environments, ranging from 0.895 to 1 (0.982 ± 0.007). Contrary to our expectation, we found no difference in total interaction dissimilarity (i.e., βWN) among urbanization levels ($F = 0.563$; $p = 0.578$). The mechanism behind the high dissimilarities of plant–herbivore interactions among sites in each of the urbanization levels was clearly the changes in species composition since the proportional contribution of βST ranged from 0.846 to 1 (0.952 ± 0.015) and βOS from 0 to 0.154 (0.048 ± 0.016). In fact, when tested for differences, the proportional contribution of βST was greater than βOS in all urbanization levels. Looking individually, the component βST represented 0.939 ± 0.067, 0.969 ± 0.051, and 0.948 ± 0.0.055 of total interaction dissimilarities in urban, rural, and wild urbanization levels, respectively (Table 1).

Since the change of species composition was the main factor affecting plant–herbivore interactions, we also shed light on the contribution of each group (*i.e.,* plants and herbivores) in terms of taxonomic dissimilarity among sites. The taxonomic beta diversity (*i.e.*, βSOR) of herbivore insects and plants ranged from 0.714 to 1 (0.918 ± 0.020) and from 0.478 to 1 (0.758 ± 0.072), respectively. As we expected, we found a greater taxonomic beta diversity in herbivore insects' group than in plants' group ($F = 28.232$; $p < 0.001$) independently of urbanization levels ($F = 1.448$; $p = 0.238$). Despite the differences in resources and conditions among

Table 1 Mean (±SD) beta diversity of interaction (βWN) and its proportional components βST (prop) and βOS (prop) among the sites within each urbanization level (wild, rural, and urban)

Beta diversity measures	Wild	Rural	Urban
βWN	0.977 ± 0.041	0.990 ± 0.033	0.979 ± 0.030
βST (prop)	0.948 ± 0.055	0.969 ± 0.051	0.939 ± 0.068
βOS (prop)	0.052 ± 0.055	0.032 ± 0.051	0.061 ± 0.068

βST (prop) represents the proportion of total beta diversity of interactions due to spatial turnover of species, while βOS (prop) represents the proportion of total beta diversity of interactions due to interaction rewiring among shared species between sites

Table 2 Mean (±SD) taxonomic beta diversity (βSOR) of both plants and herbivore insects among the sites within each urbanization level

Urbanization level	βSOR plants	βSOR herbivores
Wild	0.72 ± 0.139	0.897 ± 0.076
Rural	0.713 ± 0.121	0.938 ± 0.092
Urban	0.841 ± 0.105	0.919 ± 0.099

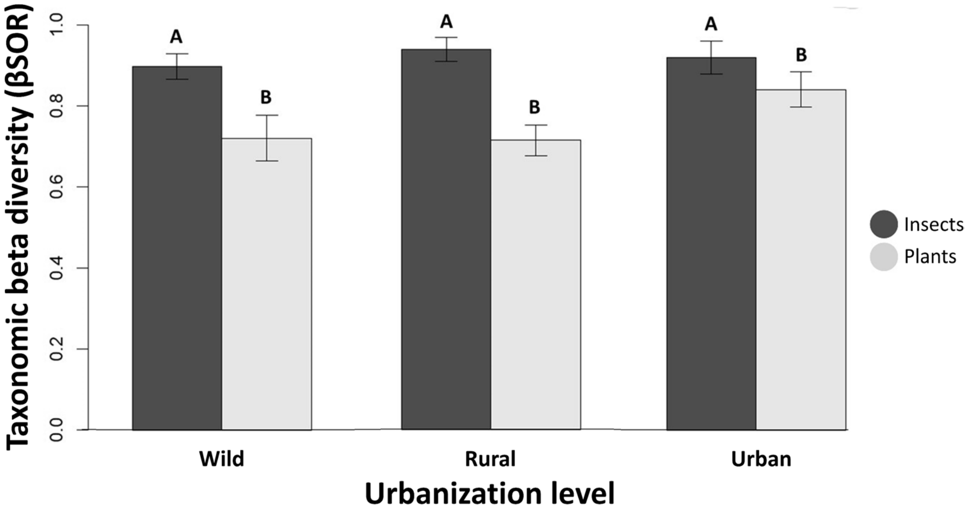
urbanization levels, the intra-sites' taxonomic beta diversity of both herbivores insects and plants does not change between them ($F = 1.747$; $p = 0.188$) (Table 2 and Fig. 2).

Discussion

Examining the beta diversity of plant–herbivore interactions across different ecosystems or spatial scales reveals the variation in species composition and their interactions (Novotny 2009). In this study, we investigate whether the beta diversity of interactions within plant–herbivore networks varies across an urbanization gradient in the Brazilian Cerrado. We expected a homogenization of plant–herbivore interactions in urban areas, due to anthropic selection of host plants and consequent attraction of specialist herbivorous insects, and that, due to the absence of natural enemies such as predators, the same species would be present and interact always with the same partners in these areas. Contrary to our initial expectations, we have found that the dissimilarity of interactions remains consistent across different levels of urbanization (urban, rural and wild) and the biologic homogenization was not observed in urban areas, indicating that these urban fragments may maintain interactions similarly dissimilar to pristine landscapes. Additionally, we found that herbivorous insects exhibit higher dissimilarity among sites than plants, regardless of the urbanization level, contributing the most to taxonomic and interaction beta diversity, corroborating our expectations. These findings are consistent with previous studies indicating that higher trophic levels exhibit greater dissimilarity of interactions than lower trophic levels (Martins et al. 2019). Our findings offer new insights into the ecological consequences of anthropogenic impacts on beta diversity of plant–herbivore interactions.

Our results revealed that the dissimilarity of interactions was similarly high across all urbanization levels and the main driving mechanism behind this was species turnover (βST). This mechanism is explained by the lack of the same interacting pair of species between evaluated sites, which affect the identity of the interaction (loss) and consequently increase the beta diversity of interactions (Belchior et al. 2023). We had hypothesized that the beta diversity of interactions would be greater in wild and rural environments compared to urban ones. Urbanization may alter the floristic composition of neotropical savanna fragments in Montes Claros, leading urban areas to be significantly distinct from natural areas (Freitas et al. 2020). However, the similar high beta diversity of interactions between herbivorous insects and plants across all levels of urbanization, from urban to

Fig. 2 Mean (bars) and standard deviation ('turned H-like' lines) of taxonomic beta diversity (βSOR) among sites within each urbanization level of both, host plants (light gray) and herbivore insects (dark gray). The letters A and B represent groups with a statistical difference



wild areas, suggests that human-altered environments can also support heterogeneous interactions systems, even with the effects of anthropogenic filtering by selecting host plant species in urban habitats. The effects on the attraction of specialist herbivorous insect species were not clear, by it seems to have occurred, as it was rare to observe the same pair of interacting species in different sites. In fact, as we demonstrated in our results, the interaction rewiring was always low, showing that despite the same pair of species rarely occur in different sites, when they make it, they tend to keep the interaction between them. This pattern has also been observed for other types of interactions (e.g., mutualistic networks) at different scales and is intensified by the geographical distance between them (Belchior et al. 2023). Thus, changes in species composition of both, plants, and herbivorous insects are the keys to understand how similar are, in terms of interactions, the studied areas. When there is a change in species composition within an ecological community, it can significantly impact the dynamics of species interactions (Anderson et al. 2006). This result leads us to consider that the drivers determining interactions between host plants and their herbivorous insects may act locally, thereby resulting in similar patterns of high species turnover in pristine and anthropized areas. This finding has implications for biodiversity conservation in urban areas, suggesting that even human-modified environments can sustain complex species plant–herbivore networks.

When considering the taxonomic dissimilarity among sites within each urbanization level, we observed a greater taxonomic beta diversity in herbivorous insects compared to plants. This result suggests that herbivorous insects contribute more to the interaction dissimilarity in plant–herbivore networks. Insect herbivores tend to have higher beta diversity in anthropic environments due to their ability to exploit a broader range of host plants, including both native and introduced species (Branco et al. 2015). However, we found that these organisms change more than plants between sites, with a similar intensity, even in more natural environments, where we would expect more specialized interactions. That is, plant composition change among sites may lead to a bigger herbivorous insects' dissimilarity. When plant community composition changes, it can disrupt the delicate balance of interactions within the network, as herbivores may lose access to their preferred host plants or encounter new plant species (Martins et al. 2019). These disruptions in plant–herbivore interactions can reverberate throughout the ecological network, potentially intensifying beta diversity at higher trophic levels (Peralta et al. 2018). As herbivorous insects respond to alterations in their host plant composition, this can affect the availability of resources for higher trophic levels, including predators and parasitoids that rely on herbivores as their prey (Martins et al. 2019). Consequently, changes in the herbivore community can lead to cascading effects on the broader ecosystem, influencing species diversity and composition at multiple trophic levels (Li et al. 2023).

Ecological networks provide a valuable framework for studying the complexity of plant–herbivore interactions (Tylianakis et al. 2010; Guimarães 2020). These networks depict the intricate web of species interactions, encompassing both direct and indirect relationships within a community. By examining the structure and dynamics of ecological networks, we can gain insights into the stability and resilience of ecosystems (Bascompte 2007). Our study provides empirical evidence that urbanization has no significantly impact the beta diversity of interactions in plant–herbivore networks in Brazilian Cerrado. Nevertheless, we found a high turnover in species composition across sites, with herbivorous insects exhibiting greater dissimilarity compared to plants. Although the results of present study indicates that networks exhibit invariant high dissimilarity between urban and wild areas, changes in species composition across the urbanization gradient may have implications for the topology of interactions performed by the species. Corroborating this, recent studies have demonstrated that the topology of plant–herbivore networks in Neotropical savannas can differ significantly between urbanized and preserved environments (Silveira and Araújo 2021; Araújo et al. 2024). Overall, these recent studies indicate that networks in urban environments are poorer in species (Araújo et al. 2024) and also less specialized in terms of structure than those in preserved environments (Silveira and Araújo 2021; Araújo et al. 2024). Therefore, the results of high dissimilarity in urban fragments should be interpreted with caution, as the topology and consequently the ecological functions of the plant–herbivore networks appear to be modified in urban environments.

In conclusion, our results suggest that while the beta diversity of plant–herbivore interactions remains similar across different levels of urbanization in the Brazilian Cerrado, there is considerable variation in species composition, with higher dissimilarity among herbivorous insects compared to plants. This implies that the effects of urbanization are more complex than initially expected, potentially influencing various trophic levels. However, they present a similar mechanism to explain interaction dissimilarities to that of wild areas. Future research could delve deeper into the impact of these changes on temporal ecosystem resilience and stability. Key areas for further study include the role of exotic and ornamental plants in urban environments and the influence of agricultural landscapes on ecological network dynamics. In addition, exploring the effects of habitat fragmentation and isolation on ecological interactions could offer more insights into the consequences of human development. Understanding these mechanisms will be crucial for developing effective conservation strategies that support the complexity of plant–herbivore interactions in urban, rural, and wild environments.

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Author contributions All authors contributed to the study conception and design. Material preparation and data collection were performed by WSA and LTS. Data analysis was performed by WSA, MMBC, LTS, MF and FSN. The first draft of the manuscript was written by WSA and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Declarations

Competing interests The authors declare that they have no financial or non-financial conflict of interest.

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