Impact of urban disturbance on soil insect communities in a Brazilian Atlantic Forest biological station

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Abstract

Urbanization constitutes a major threat to biodiversity. Understanding its effects on insect communities is relevant because they are key elements of trophic interactions, and indicators and targets of conservation. Herein, we investigated the influence of meteorological and habitat factors on the soil entomofauna in three areas with distinct levels of urbanization at the Parque Estadual da Pedra Branca, Brazil. We investigated whether community structure differs among areas with different levels of urbanization, and how changes in the environment affect soil insect community composition and distribution. We systematically monitored communities for 12 months in three areas along a gradient of anthropogenic disturbance, representing preserved secondary forest, disturbed forest and peridomicile areas. The results revealed that the degree of urbanization affects insect communities, with a strong effect of habitat factors, such as canopy cover, presence of flooded areas, quantity of fallen trunks and mean temperature. Insect abundance did not show significant differences among areas, while biomass was higher in disturbed forest than in preserved forest and peridomicile areas. Additionally, insect richness and diversity were higher in preserved and disturbed forests than in peridomicile areas, with no significant difference between preserved and disturbed forests. Our results can be used to enhance the understanding of the effects of urbanization on taxonomically and functionally diverse groups of insects, and to advise residents and urban planners about the consequences of urbanization on biodiversity and ecosystem services in urban-sylvatic interface areas.

Keywords Biodiversity · Conservation · Environmental factors · Neotropical region · Soil fauna · Urbanization

Introduction

Urbanization, the process through which urban ecosystems are created (McIntyre et al. 2001), is one of the most significant anthropogenic alterations to Earth's surface, and constitutes a major threat to global biodiversity (Fenoglio et al. 2020). Among its consequences are the increase in temperature ("urban heat-island" effect; Yow 2007), changes in rainfall and evapotranspiration patterns (Zhang et al. 2009), and the fragmentation and degradation of natural habitats (Alberti 2005). In particular, habitat fragmentation and degradation are among the most important global drivers of biodiversity loss (Fahrig 2003), leading to the homogenization of biological communities (McKinney 2006), and alterations in food web structure (Faeth et al. 2005) as well as ecological processes related to ecosystem functioning and services (Turrini et al. 2016). As urbanization is expected to increase in the next decades, forest fragments close to urban environments become increasingly important for biodiversity. Thus, understanding the effects of urban disturbance on biodiversity is crucial for effectively creating and managing forest reserves in these regions (Fenoglio et al. 2020).

Insects represent the most diverse animal group on Earth, hence their relevance as indicators and targets of conservation is well-established (McGeoch 1998; Uehara-Prado et

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al. 2009). They are key elements of terrestrial trophic interactions, providing essential ecosystem services of global relevance, such as pollination, biological pest control, nutrient cycling, and decomposition (Losey and Vaughan 2006), generating considerable economic and cultural benefits for society. Urbanization has been shown to promote large changes in arthropod communities (McKinney 2006; Groffman et al. 2014), altering their structure and species interactions, which may often affect important abiotic and biotic processes (e.g., nutrient cycling, decomposition; McIntyre et al. 2001).

Studies have indicated contrasting effects of urbanization on arthropod communities and a general pattern remains inconclusive (Fenoglio et al. 2020). While some studies have indicated a decrease in species richness with increasing urbanization levels (McKinney 2008; Martinson and Raupp 2013; Fenoglio et al. 2020), others have shown that urbanization can favor the occurrence of non-indigenous (McKinney 2002; Cadotte et al. 2017), opportunistic, and generalist species (Faeth et al. 2005). Moreover, the effects of urbanization on the terrestrial biota may vary among cities with different climates, geographies, and land uses (Faeth et al. 2011; Aronson et al. 2016). In the neotropics, several studies have explored how insect communities of certain orders are susceptible to habitat changes related to the disturbance of natural environments and their conversion into pastures or monocultures (e.g., Nichols et al. 2007; Larsen et al. 2008; Neves et al. 2010; Silva et al. 2010; Filgueiras et al. 2011; Liberal et al. 2011). However, few studies have described temporal and spatial changes in insect communities along urban-sylvatic or rural-sylvatic gradients, and they are often limited to specific families or orders (e.g., Aguiar and Gaglianone 2012; Korasaki et al. 2013; Rocha-Ortega and Castaño-Meneses 2015).

The Pedra Branca State Park, or Parque Estadual da Pedra Branca (PEPB; INEA-RJ 2013), in the city of Rio de Janeiro, Brazil, is one of the most important urban forests in the world and a potentially ideal system to explore urbanization impact on arthropod communities. Legally established in 1974, it is the largest forest reserve located in an urban area in Brazil, and one of the largest remnants of the Atlantic Forest (Dean 1995), a biome that supports one of the highest degrees of species richness and rates of endemism on the planet (Myers et al. 2000). The PEPB is characterized by secondary forest formed after intense exploration of coffee, citrus cultivation and coal production from the mid-nineteenth century to the early twentieth century (Oliveira 2010). It occupies parts of 17 neighborhoods and 10% of the total municipal area, assuming a great conservation and social importance (Ribeiro and Ribeiro 2016), and crucial roles in climate regulation and ecosystem function in the region (INEA-RJ 2013). However, it faces a large array of pressures derived from disordered and rapid urbanization, as well as irregular activities within and around the park (e.g., housing, small agricultural practices; Guimarães and Pelin 2015). Moreover, despite its relevance for the conservation of taxonomic and functional diversity in an urban area, very few studies highlight its arthropod diversity (e.g., Castanheira et al. 2016; Avelino-Capistrano et al. 2017).

Given the relevance of conserving forests in fragmented landscapes, we measured and evaluated the influence of meteorological variables and habitat features on the soil entomofauna in three areas with distinct levels of urbanization (i.e., preserved secondary forest, disturbed forest and peridomicile areas) in the surroundings of the PEPB. We hypothesize that the urbanization process affects the composition and structure of the soil insect community (i.e., diversity, richness, abundance and biomass) in a gradient of anthropogenic disturbance. Our results can be used to enhance the understanding of the effects of urbanization on taxonomically and functionally diverse groups of insects, and to advise residents and urban planners about the consequences of urbanization on biodiversity and ecosystem services in urban-sylvatic interface areas.

Materials and methods

Study area. The PEPB is located in the West Zone of Rio de Janeiro municipality (Fig. 1). It occupies an area of 12,500 ha, corresponding to 11% of the city area, and has a humid tropical climate. The study was carried out in three areas at FIOCRUZ Atlantic Forest Biological Station (Estação Biológica FIOCRUZ Mata Atlântica - EFMA), which is part of the buffer zone of PEPB, including preserved areas of the park (Fig. 1). The preserved forest area (22°56'47", S 43°25'07" W) is located approximately 1 km from human habitations and close to or above 100 m high, which defines the lower limits of the PEPB. This area presentes canopy heights ranging from 10 to 40 m and irregular slopes ranging from flat to steep. The disturbed forest area (22°56'28", S 43°24'34" W) has vegetation in regeneration process with different stages of ecological succession. This area is located between the peridomicile area and the preserved forest, with canopy heights ranging from 6 to 20 m and flat slope. The peridomicile area (22°56'18"S, 43°24'11"W) is located in the backyards of the houses of residents of the locality, and has a low canopy ranging from 6 to 15 m, predominance of shrubs and small trees, presence of flooded areas and slopes ranging from flat to moderate.

Field sampling. Sampling was conducted monthly from January to December 2019. It was performed on nine transects (three in each area) of ten sampling stations each, equidistant in 10 m, totaling 90 sampling stations. The pitfall
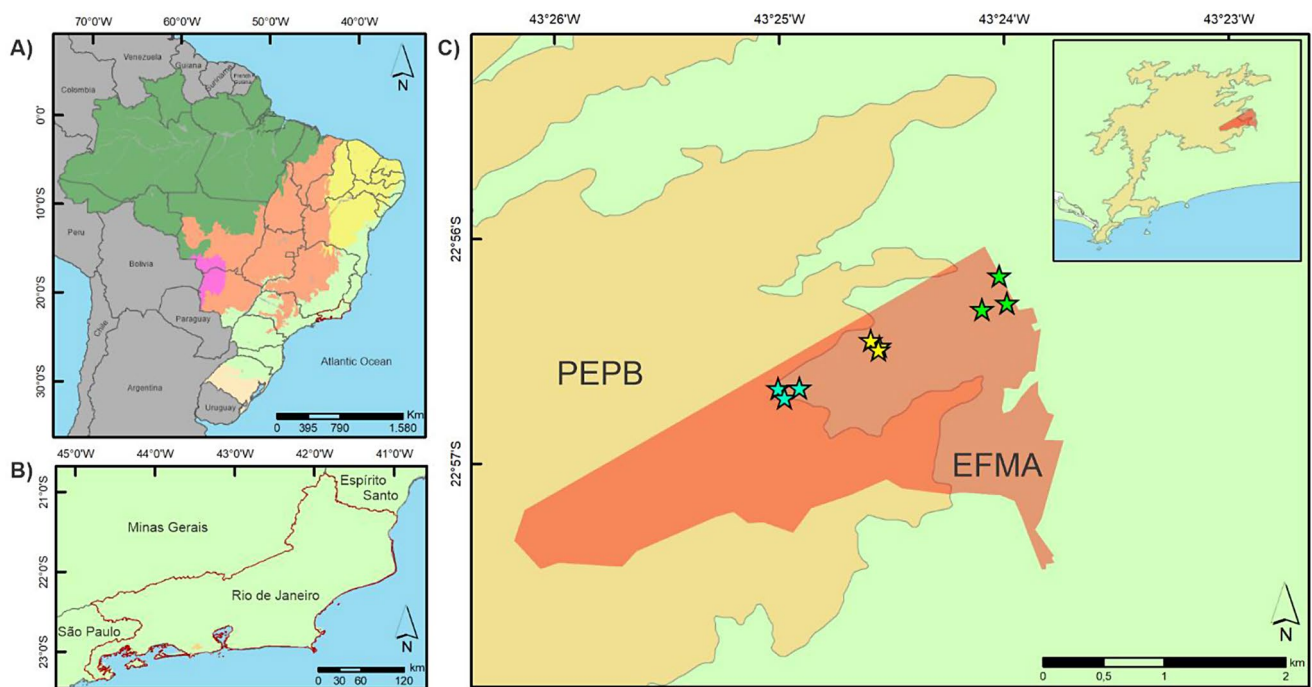
Fig. 1 Geographic location of the study area and sampling transects. (**A**) Location of Brazil in South America, with internal divisions by states (Rio de Janeiro state margined in red) and biomes: Amazon (dark green), Cerrado (orange), Pantanal (pink), Caatinga (yellow), Atlantic Forest (light green), and Pampa (light beige). (**B**) Location of Rio de Janeiro State (margined in red) within the Atlantic Forest, and of Pedra Branca State Park (PEPB; highlighted in dark beige) within the state. (**C**) Geographic location of PEPB (highlighted in dark beige) and FIOCRUZ Atlantic Forest Biological Station (EFMA; highlighted in red) in Rio de Janeiro state. The largest map shows the locations of the sampling transects in the intersection of PEPB and EFMA. Light blue stars: preserved secondary forest; yellow stars: disturbed forest; green stars: peridomicile areas

traps were placed in the soil for 72 h once a month. Traps consisted of plastic cups of 10-cm diameter and 12-15-cm depth, filled with soapy water. Collected material was then fixed in 70% ethanol and transported to the laboratory for identification.

Due to the high diversity and complexity of the Neotropical fauna, specimens were identified to the family level. Many studies have shown that analyses at this level are adequate to evaluate the effects of changes in community structure (e.g., Somerfield and Clarke 1995; Chapman 1998). We used the keys provided in Triplehorn and Johnson (2011) and Rafael et al. (2012), and Zeiss Stemi SV-11 stereomicroscopes for the identification.

After identification, the insect samples were washed in running water using a sieve, placed in a petri dish separated by sampling point and dried at temperatures between 40 °C and 60ºC in a laboratory stove (Roni Alzi Vidros Científicos Ltda, model RA-40) according to the size of the insects. The dried insects were weighed on a digital balance (Bel Engineering brand, model Mark N1203). Biomass was estimated per sampling point in grams.

Meteorological variables and habitat measurements. To measure the effect of habitat on the soil insect family structure across communities, we used the following independent variables: percentage of canopy cover, number of fallen tree trunks and presence of flooded areas (following Freitas et al. 2002). These variables were obtained at each sampling station, in August 2019. We also tested the effect of meteorological variables on the soil insect family richness, family diversity, total abundance and mean biomass. We used the following meteorological variables: average temperature during the 30 days prior to each collection, accumulated rainfall during the 30 days prior to each collection, and accumulated rainfall during the seven days prior to each collection. These data were obtained from a meteorological station located in the study area (Instituto Nacional de Metereologia Automatic Station RJ A636 – Jacarepaguá, Rio de Janeiro).

Statistical analysis. To display the dissimilarity among the three areas studied, we used non-metric Multidimensional Scaling (nMDS) analysis, based on Bray-Curtis dissimilarity matrices of total abundance. Prior to the analysis, total abundance values were square root transformed and a dummy variable was added to the data. The stress value in nMDS analysis was too high (0.19) when including the effects of monthly sampling (Supplemental Material S1, Fig. S1). This fact can lead to potential misinterpretation of the ordination (Clarke 1993). Therefore, we included

only the results with the polled dataset by transects for this analysis. Additionally, a permutational analysis of variance (PERMANOVA) was used to test whether differences among areas were statistically significant, using $\alpha = 0.05$. Due to the low number of possible unique permutations (30 permutations), pairwise *post-hoc* comparisons between areas were not conducted. To assess which taxa/function contributed most to the dissimilarity between treatments, Similarity Percentage Analysis (SIMPER) (Clarke 1993) was employed.

To evaluate the relationship between habitat variables and the community composition of insects (family structure), the BIOENV procedure was used (Clarke and Ainsworth 1993) based on a Spearman rank correlation between the Bray–Curtis similarity matrix and habitat data. Total abundance values were transformed following the same procedures described above. Habitat variables (i.e., percentage of canopy cover, number of fallen tree trunks and flooded areas) were $\log_e(x+1)$-transformed prior to the analysis, and a principal component analysis (PCA) was also used to display the relation between habitat variables and sampling areas.

To evaluate the effects of temperature and rainfall on family diversity (i.e., Shannon index), family richness, total abundance, and mean biomass (g), Bayesian generalized linear mixed-effect models (GLMMs) were used. Because habitat variables were assumed to be constant under the course of this study, they were not included in this analysis. For all metrics, the data of each sampling station were pooled by area and month. GLMMs assuming negative binomial distributions were used for richness and abundance, and a Gaussian distribution was used for diversity and biomass, which were the best fitted models for each case (Supplemental Material S2, Tables S1–S4). The random effect of sampling site was included in the analysis and both additive and interaction models were considered. Gaussian weakly informative priors ($\mu = 0$, $\sigma = 2$) were used for the fixed effects (i.e., temperature, rainfall, sampling area and interaction terms). For the φ parameter in the negative binomial regression, a gamma distribution with both scale and shape parameters equal to 0.01 was used, and a Student-t distribution was used for the random effects of area ($\mu = 0$, $\sigma = 2.5$ and $df = 3$). Model selection was performed based on leave-one-out cross validation (LOO) information criteria and LOO weights, as described in Vehtari et al. (2017). Models were fitted using Markov Chain Monte Carlo (MCMC) methods with a Hamiltonian sampler algorithm. Four independent chains were used in parallel with initial 5000 iterations for adaptation, followed by 10,000 iterations. Evidence ratios (ER) were used to test the one-sided hypothesis that the explanatory (independent) variables were higher or lower than zero. All statistical analyses were conducted in the R environment (R Development Core Team 2022) with the *vegan* (Oksanen et al. 2012) and brms (Bürkner 2017) packages.

Results

A total of 105 insect families was identified among 14,187 individuals considering all areas (Supplemental Material S3, Table S5). Hymenoptera was the most abundant order across all three areas, followed by Coleoptera and Diptera (Supplemental Material S3, Table S5). The families Formicidae (Hymenoptera; $n = 12{,}531$), Scarabaeidae (Coleoptera; $n = 439$) and Gryllidae (Orthoptera; $n = 300$) were the most abundant (Fig. 2). April recorded the highest insect abundance, however, the highest relative frequency of the number of families was observed in January (Fig. 3).

The PERMANOVA indicated significant differences in the structure of insect communities between areas (Table 1; Fig. 4). Based on the SIMPER analysis, Periscelididae, Sciaridae, Culicidae and Sphaeroceridae (Diptera) were the families that contributed most to the dissimilarity between preserved forest and peridomicile areas (9% of the cumulative dissimilarity) (Supplemental Material S4, Tables S6–S8). Meanwhile, Labiduridae (Dermaptera), Apidae (Hymenoptera) and Aphididae (Hemiptera) were the families that contributed most to the dissimilarity between peridomicile and disturbed forest areas (8% of the cumulative dissimilarity).
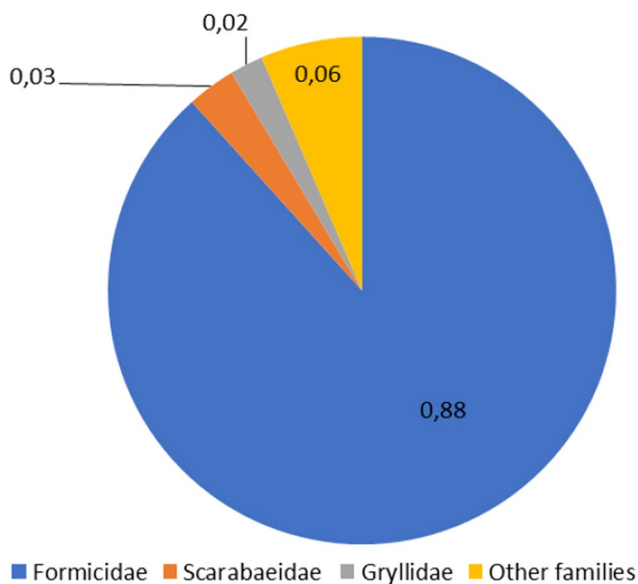

Fig. 2 Relative frequencies of Formicidae (Hymenoptera), Scarabaeidae (Coleoptera), Gryllidae (Orthoptera) and all other insect families combined from January to December 2019 in the peridomicile, disturbed forest and preserved secondary forest at Fiocruz Atlantic Forest Biological Station, Rio de Janeiro, Brazil
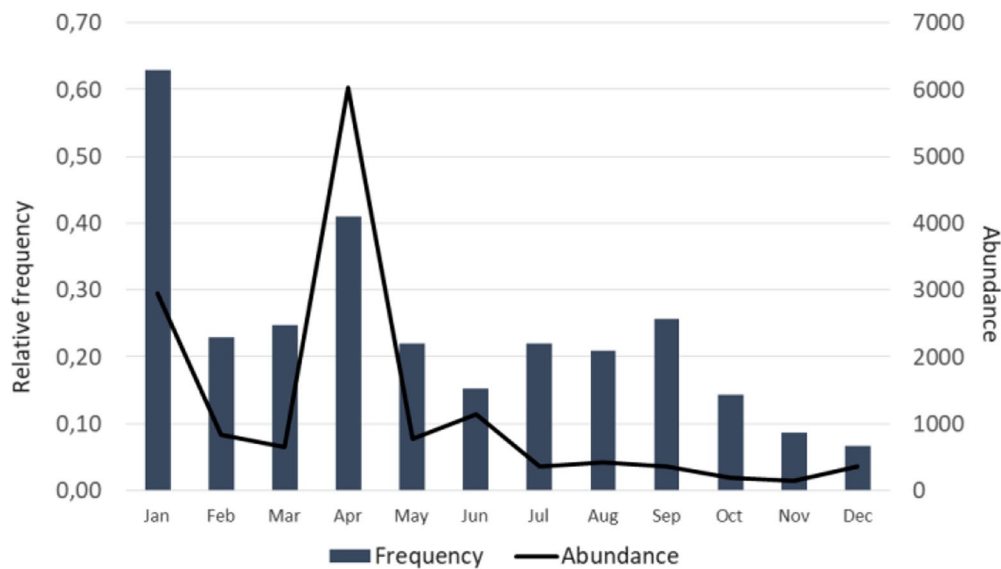
Fig. 3 Total insect abundance and relative frequency of the number of insect families from January to December 2019 in the peridomicile, disturbed forest and preserved secondary forest at Fiocruz Atlantic Forest Biological Station, Rio de Janeiro, Brazil

Table 1 Results of the Permutational analysis of variance (PER-MANOVA) of insect families among areas at Fiocruz Atlantic Forest Biological Station, Rio De Janeiro, Brazil (*Df*: degree of freedom; *MS*: mean squares; *Pseudo-F ratio*; *Pseud-R²*; and *P-value*)

Factor	Df	MS	Pseudo-F	Pseudo-R^2	*P*-value
Sampling area	2	0.27	2.00	0.40	0.011
Residuals	6	0.13		0.60	
Total	8			1.00	

All habitat variables used (i.e., canopy cover, number of flooded areas and fallen tree trunks) in the BIOENV analysis were correlated with the insect family structure ($\rho = 0.58$, $P = 0.02$). The preserved forest had a higher number of flooded areas and denser canopy cover than the other areas,

while the peridomicile areas had a higher number of fallen tree trunks than the forest areas (Fig. 4B).

For family richness, the best fitted model included the effects of sampling area and mean temperature. The results from model selection are provided in Supplemental Material S4, Tables S6–S8, and parameter estimates are provided in Supplemental Material S5, Tables S9–S12. Our results indicate that the number of families increased as mean temperature rose (ER > 1000) and that there is higher richness in the preserved forest when compared with the peridomicile (ER = 68) (Fig. 5A).

For abundance, the best selected model also included the effects of sampling area and mean temperature. Like for richness, abundance increased as mean temperature rose
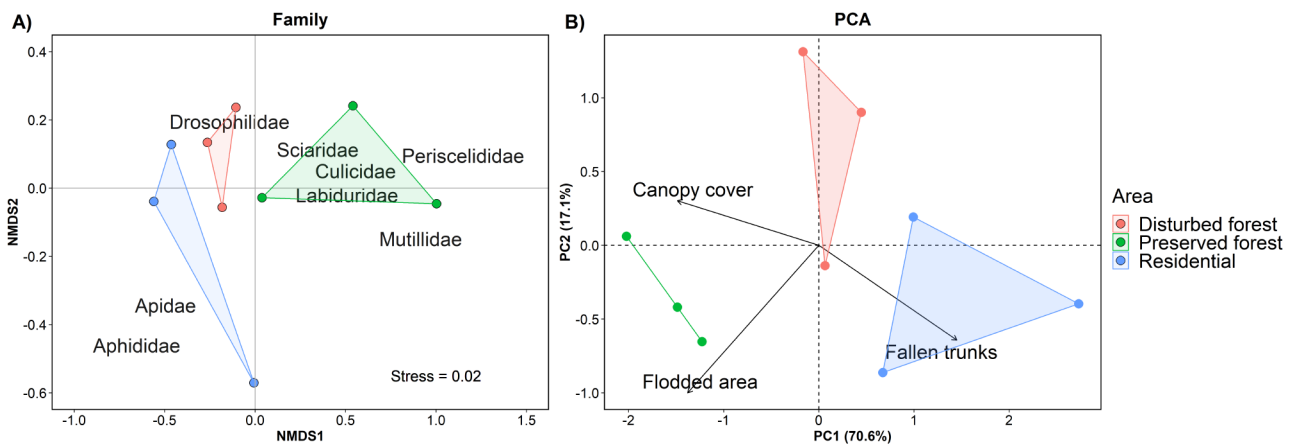

Fig. 4 Non-metric multidimensional scaling (*nMDS*) ordination plot comparing the sampling areas (**A**) and Principal Component Analysis (PCA) of habitat variables (**B**) at Fiocruz Atlantic Forest Biological Station, Rio de Janeiro, Brazil. Each point represents a sampling tran-

sect, and they are linked by area using Minimum Convex Polygons. The family names in (**A**) are the most important taxa contributing to the dissimilarity among areas based on the SIMPER analysis. Arrows in (**B**) represent the contribution of each variable in the PCA
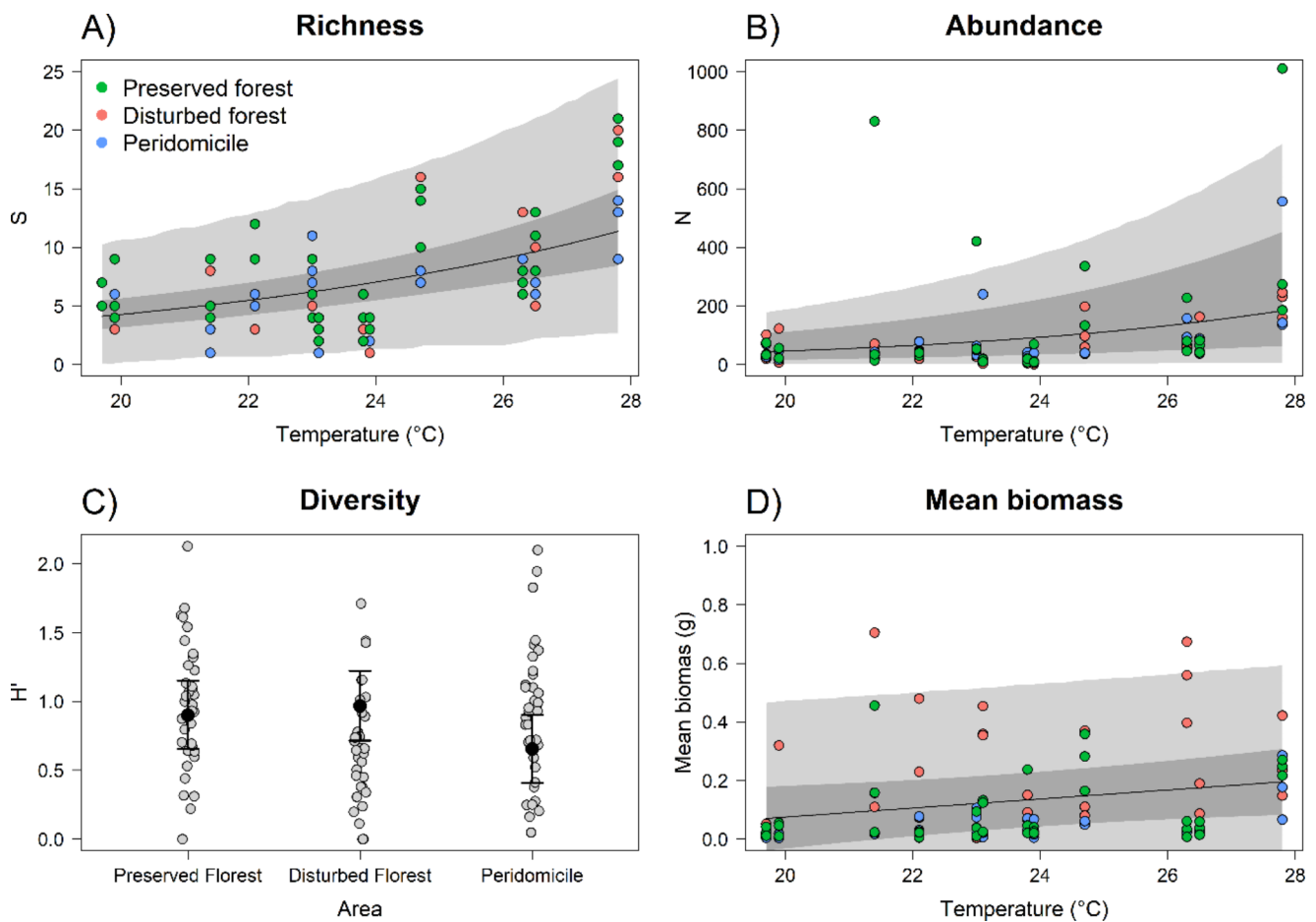
Fig. 5 Plots of the best fitted generalized linear mixed-effect models on insect parameters at Fiocruz Atlantic Forest Biological Station, Rio de Janeiro, Brazil. (**A**) Effect of mean temperature on family richness; (**B**) effect of mean temperature on total abundance; (**C**) effect of the sampling area on family diversity (Shannon index); and (**D**) effect of mean temperature on mean biomass (g). Each point in the plot represents one sampling month in each area. Solid lines represent the mean estimated effect, and error bars and gray shaded areas represent 95% predicted and fitted confidence intervals

(ER > 1000). However, differences in abundance between disturbed and preserved forest areas, and between peridomicile and preserved forest areas, were weakly supported (ER = 4.1 and 5.3, respectively) (Fig. 5B).

For family diversity, the best selected model included only the effect of the sampling area, with lower diversity in the peridomicile area (ER = 14.8), but no differences between preserved and disturbed forest areas (ER = 2.1) (Fig. 5C). Biomass, like richness and abundance, is also under positive effect of the temperature (ER = 43). Moreover, biomass is higher in the disturbed forest in comparison to the preserved forest (ER = 141) and peridomicile (ER = 587) areas (Fig. 5D).

Discussion

Our results indicate that the degree of urbanization and the consequent differences in habitat factors and meteorological variables had a clear effect on insect communities. Their abundance, biomass, richness, and diversity displayed an idiosyncratic and non-progressive response to the spectrum of studied areas, from the preserved forest to the peridomicile.

Habitat and meteorological factors

Our results show that the habitat and meteorological features selected are highly associated with the insect communities in the areas with distinct levels of urbanization. Changes to these features can alter food resources available (e.g., dead and dying wood, stressed foliage, and supplies of litter, blossoms, seeds, and fruits) and physical conditions (e.g., microclimate, microsites or structurally diverse vegetation),

which in turn greatly affect insect distribution and habitat selection (Peck and Forsyth 1982; Bouget and Duelli 2004; Korasaki et al. 2012). For instance, canopy cover has been linked to the performance of numerous species, leading to changes in distribution patterns, abundance, and diversity (Werner and Glennemeier 1999; Skelly et al. 1999; Grether et al. 2001; Muscolo et al. 2014). Reduction of coverage due to the mortality of single trees, or even all trees in extensive stands, might result in the penetration of more light to lower layers, decreasing soil moisture, increasing daytime air and soil temperatures (Scharenbroch and Bockheim 2008; Muscolo et al. 2014) and thus, also shifting microclimate and causing trophic changes.

The preserved forest area had a higher number of flooded areas and higher percentage of canopy cover than the other areas, creating an ideal environment for the Dipteran families that contributed to the observed dissimilarity between areas. The families Sciaridae, Culicidae, Sphaeroceridae and Periscelididae are strongly related to warm, humid tropical areas with forest environments and dense canopy coverage (Battirola et al. 2007; Costa et al. 2010; Lega et al. 2017; Borkent et al. 2018; Brown et al. 2018). However, in particular for Diptera, previous studies associating canopy cover with diversity measures (i.e., richness, diversity and abundance) have shown contrasting results, varying according to the family studied. While some have suggested that a denser canopy cover reduces herb cover and herb diversity, negatively affecting abundance and diversity (e.g., Woodcock et al. 2003; Scherber et al. 2014), others have indicated a positive association between Diptera diversity and greater tree density and canopy shading (Carlson et al. 2016; Amorim et al. 2022).

This strongly contrasts with the peridomicile area, which was dissimilar to the forest areas, mostly due to the presence of Labiduridae (Dermaptera), Apidae (Hymenoptera) and Aphididae (Hemiptera). Especially for the family Apidae, its abundance in an urban tropical area could be explained due to its association with flowering plants for resources (Silva et al. 2017) and with large proportions of beneficial habitats rich in forage and nesting resources, such as gardens, as well as remnant patches of native habitats (Matteson et al. 2013; Pellissier et al. 2013; Kaluza et al. 2016). It is well known however, that nesting strategy seems to be of great importance for this group, since most bee species are ground-nesting and negatively influenced by urbanization, while cavity-nesting species are positively influenced by this process (Wojcik 2011; Hinners et al. 2012; Cardoso and Gonçalves 2018).

The diversity in the three areas was mostly represented by Diptera, followed by Coleoptera and Hymenoptera. These groups have also been recovered as the most abundant at the ground level in other studies on Neotropical preserved forests (e.g., Amorim et al. 2022). The order Coleoptera showed lower diversity but slightly higher abundance than Diptera in the three areas. The composition of the Coleoptera families did not vary strongly between areas, and was mostly represented by the families Scarabaeidae, followed by Curculionidae and Chrysomelidae. The predominance of Scarabaeidae could be explained by their foraging behavior close to the ground (Davis et al. 1997), leading to easier capture considering the sampling method used, i.e., pitfall traps. Additionally, the abundance of this family was higher in preserved areas and gradually reduced towards disturbed and peridomicile areas, which can be explained by the public illumination in the latter. It can act as a light trap affecting nocturnal species abundance (e.g., Frankie and Ehler 1978), promoting changes in orientation, foraging patterns, reproduction, migration and communication (Longcore and Rich 2004; Korasaki et al. 2012). We also noted a slight increase in Erotylidae abundance in disturbed and peridomicile areas, which can be associated with the higher availability of fallen tree trunks in those areas than in the preserved area, which could provide plant and fungal matter for the insects to feed on (Leschen and Buckley 2007). Surprisingly, ground beetles (family Carabidae), a major group of soil fauna (Vennila and Rajagopal 1999), were rarely recovered in our samples. This can be due to the transparent plastic cups used as traps because these beetles have been shown to be more attracted to white or yellow (Buchholz et al. 2010) or even to the trap design used (Digweed et al. 1995). Hymenoptera was the most abundant order in the three areas and was predominantly represented by the family Formicidae. This family is known for using the substrate of tropical forests for nesting or foraging. Additionally, the leaf-litter stratum of these forests exhibits great abundance of ant species, which are associated with a variety of available locations for nest building (Fernandes et al. 2012).

As ectotherms, insects are highly responsive to temperature (McIntyre et al. 2001). Although some insect taxa might benefit from rising temperatures (Termaat et al. 2019; Baker et al. 2021), the thermal adaptation hypothesis suggests that in the hot, aseasonal tropics, species are operating at, or close to, their thermal optimum. Therefore, small increases in temperature can have drastic effects on ectotherm abundance (Deutsch et al. 2008; Angilletta 2009; Sunday et al. 2014; Woon et al. 2019). However, our results showed a positive effect of temperature along the urbanization gradient on measures of insect abundance, richness and biomass, which has also been demonstrated in previous studies (e.g., Hallmann et al. 2017; Uhler et al. 2021; Welti et al. 2022). Conversely, rainfall showed no effect on any of the insect parameters measured. In previous works, there was no clear relationship between rainfall and arthropod communities in tropical areas. While some indicated a general positive

effect of increased rainfall on abundance, biomass and richness (Wolda 1978; Tanaka and Tanaka 1982; Liberal et al. 2011), others showed contrasting patterns within and between guilds (Denlinger 1980; Weissflog et al. 2018), or no effect at all (Blanche 2000; Veldtman & McGeoch 2003).

Richness, diversity, abundance, and biomass

In our study, we evaluated the effects of urbanization not only on insect richness and abundance, which is the common approach (McDonnell and Hahs 2008), but also on biomass and diversity. The results showed that urbanization had a negative effect on both insect richness and diversity, indicating that communities were poorest and least diverse in the most urbanized landscapes. This agrees with the increasing disturbance hypothesis (Gray 1989), which states that arthropod richness declines with an increasing level of urbanization, with the lowest values in the urban core (McKinney 2008; Martinson and Raupp 2013). Because temperature had a positive effect on insect abundance, richness and biomass in our study area, this negative effect of the urbanization may be a consequence of resource reduction or of habitat structure alteration, not of the "urban heat island effect" (Yow 2007). Previous studies that used Neotropical ants (Rocha-Ortega and Castaño-Meneses 2015), beetles (Korasaki et al. 2013) and semiaquatic bugs (Cunha et al. 2022) as models to assess the effect of urbanization on insect communities varied in their results. The first study found higher species richness in sites with intermediate disturbance and similar diversity indices among sites across an urbanization gradient, which was explained by the disturbance heterogeneity model (Porter et al. 2001). This model states that moderate levels of human disturbance promote the coexistence of different species, leading to increased habitat diversity and higher biodiversity indices. In turn, our results are in accordance with the second and third studies, which found the lowest richness in the most urbanized areas, although they did not evaluate diversity. This corroborates studies developed in temperate regions, where the most urbanized areas were found to harbor poorer and less diverse bee and butterfly communities, for example (Tzortzakaki et al. 2019; Herrmann et al. 2023).

In our study, abundance did not show significant differences between areas, different from what has been found for beetles (Korasaki et al. 2013). However, for beetles, this was attributed to the dissimilar efficiency of different bait types used in the traps along the urbanization gradient, in contrast to the non-baited traps of our study. The effects of urbanization on the abundance of Neotropical ants and semiaquatic bugs have never been investigated

(Rocha-Ortega and Castaño-Meneses 2015; Cunha et al. 2022), while in temperate areas the effects vary depending on the taxon and functional group studied (Brasil et al. 2023; Fang et al. 2023). Finally, we found that insect biomass was higher in the disturbed forest than in the preserved forest and in the peridomicile areas. Biomass is related to productivity but has been poorly investigated along gradients of urbanization (Beninde et al. 2015), although Uhler et al. (2021) showed a large decrease in insect biomass from semi-natural to urban environments in a temperate area.

Conclusion

Urbanization had a clear effect on the soil insect communities of the border areas of the studied forest reserve. A clear gradient in the community structure was observed from the preserved forest to the peridomicile areas. Although the results differed among the community parameters assessed, richness and diversity clearly decreased along the urbanization gradient, with higher indices in the more preserved area than in the disturbed area. This study highlights the importance of monitoring and managing areas in urban-sylvatic interface areas to maintain biodiversity and, consequently, human access to invaluable ecosystem services provided by forest reserves in urban centers, such as the PEPB.

RGB, KCS. Software: MK, RG, MVPS. Supervision: FFFM, JMSR, RG, MVPS. Validation: FFFM, RG, MVPS. Visualization: MK, KCS. Writing - original draft: KCS, MVPS. Writing - review & editing: FFFM, JMSR, MK, RGB, RG.

Data availability Data is provided within the manuscript and supplementary information files.

Declarations

Competing interests The authors declare no competing interests.

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