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# Effect of green spaces and floral composition on the community of flower visitors in a piedmont city in Argentina

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**Abstract.** In cities, human activities transform the native composition of plant communities into a mixture of native and exotic species. This new plant community shapes the composition of floral visitor communities persisting in urban environments. Recognizing the value of the diversity of urban pollinator communities can help quantify the conservation significance of cities. The aims of this work are: 1) to understand the effect of urbanization on flower visitors by comparing pollinator communities in natural and urban landscapes; 2) to quantify the influence of floral resources on floral visitors, and 3) to assess the impact of green spaces in urban environments on the composition of the flower-visitor community. We explored plant communities and floral visitors in five urban green spaces in a low urbanized city in Tucumán, Argentina, and in five natural protected areas located adjacent to the city. We found a greater diversity of flowering plants in the urban environment; mainly represented by exotic plant species and a lower diversity of the pollinator community than in the natural landscape. Pollinator diversity also showed a positive association with plant richness, while it was negatively affected by the proportion of exotic plants and the increase in flower density. Within city, distance to green spaces negatively affects the diversity of pollinators. Our results show that floral visitor communities in urban environments are less diverse than those in natural sites. Therefore, the composition of the plant community in the city, mainly exotic seems to affect communities of urban floral visitors. Our findings suggest that cities without intensive urbanization, and that still share many species with the closest natural space, may focus on urban planning that considers development alternatives that contribute to the conservation of pollinators.

[Keywords: urban greenspaces, flower density, exotic plants, potential urban pollinators, urban environment]

**Resumen. Efecto de los espacios verdes y la composición floral en la comunidad visitantes florales en una ciudad del pedemonte de Argentina.** En las ciudades, las actividades humanas transforman la composición nativa de las comunidades de plantas en una mezcla de especies nativas y exóticas. Esta nueva comunidad vegetal influye en las comunidades de visitantes florales en entornos urbanos. Reconocer el valor de la diversidad de polinizadores urbanos puede ayudar a cuantificar la importancia de la conservación en las ciudades. Los objetivos de este estudio son: 1) entender el efecto de la urbanización en los visitantes de flores comparando las comunidades de polinizadores en paisajes naturales y urbanos; 2) cuantificar la influencia de los recursos florales en los visitantes, y 3) evaluar el impacto de los espacios verdes en la composición de la comunidad de visitantes florales. Se exploraron comunidades de plantas y visitantes florales en cinco espacios verdes urbanos en una ciudad poco urbanizada de Tucumán, Argentina, y en cinco áreas naturales protegidas adyacentes. Se encontró una mayor diversidad de plantas con flores en el entorno urbano, principalmente especies exóticas, y una menor diversidad de polinizadores en comparación con el paisaje natural. La diversidad de polinizadores mostró una asociación positiva con la riqueza de plantas, pero se vio afectada negativamente por la proporción de plantas exóticas y el aumento de la densidad floral. Dentro de la ciudad, la distancia a los espacios verdes afecta negativamente la diversidad de polinizadores. En conclusión, las comunidades de visitantes florales en áreas urbanas son menos diversas que en las áreas naturales, y la composición vegetal urbana, mayormente exótica, influye en estas comunidades. Se recomienda que las ciudades con baja urbanización, que aún comparten especies con áreas naturales, adopten estrategias de planificación urbana que promuevan la conservación de los polinizadores.

[Palabras clave: espacios verdes urbanos, densidad de flores, plantas exóticas, potenciales polinizadores urbanos, ambiente urbano]

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# Introduction

The Anthropocene is characterized by human impact on different landscapes at the global scale, the growth of the human population and an increasing pressure on natural resources. This leads to a transformation and loss of natural environments, which are being replaced by human infrastructure such as urbanization, agriculture, and roads, among others (Waters et al. 2016). Urbanization represents an impact concentrated on relatively small areas, in contrast to other activities (e.g., agriculture affects larger areas compared to urbanization). Urbanization produces a variety of effects, including loss, isolation, fragmentation and pollution of natural environments, as well as the introduction of exotic species (Grimm et al. 2008). As a product of urbanization, different human activities related to the maintenance of public and private green spaces seek to compensate for natural or seminatural green spaces, resulting in a new floristic community composed of a mixture of native and exotic species (Knapp et al. 2012). In fact, people in cities might intentionally select plants with certain traits based on their aesthetic value (Lindenmann-Matthies et al. 2010), such as flower color and symmetry, which could affect interactions between plants and pollinators (Giurfa et al. 1999).

Although urbanization serves as a filter for specific groups of floral visitors (Amado De Santis and Chacoff 2020; Bates et al. 2011; Wenzel et al. 2020), many studies assessing the richness and abundance of wild pollinators concluded that diverse pollinator groups persist in various cities, supported by these emerging floristic communities (Baldock et al. 2015; Banaszak-Cibicka et al. 2018; Geslin et al. 2016; Hall et al. 2017; McFrederick and LeBuhn 2006). This led to reconsidering the idea of cities, traditionally seen as desert islands, as environments where a great diversity of pollinators can inhabit (Hall et al. 2017). In this way, cities could play a potential role in pollinator conservation by providing refuges and alternative foraging places to many floral visitors (Fischer et al. 2016; Hall et al. 2017). The abundance and diversity of bees and butterflies are more influenced by local landscape characteristics, such as the availability and quality of floral resources and nesting sites, than by the intensity of urbanization (Ahrné et al. 2009; Dylewski et al. 2019, 2020; Kearns and Oliveras 2009; McFrederick and LeBuhn 2006). Despite the high richness of urban plant communities in urban areas, flower visitors only use a small proportion of the resources compared to other human-modified landscapes, such as agricultural landscapes (Lowenstein et al. 2019; Theodorou et al. 2020). Baldock et al. (2019) observed that native plants were the main foraging resources for most urban pollinators, while exotic plants became an alternative resource in the absence of native plants. It was reported that in urban landscapes, there is a prevalence of generalist over specialist pollinator species, since the former have greater plasticity for the use of resources and hence can use the non-native plant species that are common in cities (Normandin et al. 2017).

Identifying how different traits of flowering plants in urban areas impact flower visitors and recognizing the potential value of a diverse ensemble of pollinators in cities can contribute to quantifying the conservation significance of urban environments and to managing them to increase efforts for pollinator conservation. A distinctive feature of floral resources in urban environments is their clustered distribution, a result of human intervention in landscape management. The distribution of floral resources can drive the distribution of pollinators, affecting their richness and abundance in cities (Graffigna et al. 2023; Persson et al. 2022a). Therefore, urban green spaces, such as private gardens, parks and other public green spaces, can sustain a high density of wild bees and other potential pollinators (Baldock et al. 2019; Banaszak-Cibicka et al. 2018; Dylewski et al. 2020; Geslin et al. 2016). In this way, floral and nesting resources determine the quality of the most suitable green spaces for urban pollinators (Baldock et al. 2019; Dylewski et al. 2020; McFrederick and LeBuhn 2006; Normandin et al. 2017). However, most studies on urban pollination only consider a subset of pollinator guilds and are limited to comparisons with other types of land use (urban, natural, agricultural). Meanwhile, the importance of green spaces, such as squares and parks (Geslin et al. 2016), as well as the distance to them and the composition of the floral community on pollinator diversity, have recently become the focus of research (Persson et al. 2022a; Persson et al. 2022b).

In the Tucumán province, in northwestern Argentina, there is an urban conglomerate (Gran San Miguel de Tucumán, hereafter GSMT) right next to a natural area of subtropical cloud forest (Sierra de San Javier

Provincial Park, hereafter SSJ). Together, they constitute a dynamic urban-natural system that experienced notable changes in land use throughout history (Gutiérrez Angonese 2015). The city of Yerba Buena, located in the western region of the GSMT at the piedmont of SSJ, had very few residences until the mid-19th century, which were mainly related to rural activity. Gradually, the city evolved towards urbanization in patches with an accelerated modification of the landscape and the development of the main access routes (Haedo et al. 2010). In recent decades, the city of Yerba Buena experienced rapid growth (Statistics Directorate of the Tucumán Province 2020). Urbanization in this city is characterized by low population density, with numerous residential homes that have large green spaces, and where native vegetation is present mainly in vacant lots, squares, urban trees and plants that were integrated into the backyards (Haedo et al. 2010).

The goals of this study are 1) to understand the effect of urbanization on potential pollinators by comparing the community of flower visitors between natural and urban landscapes; 2) to quantify the effect of floral resources in terms of density, species richness, diversity, and status (native or exotic plant species) on floral visitors, and see whether these variables differ between urban and natural environments, and 3) to assess whether urban green spaces work as refuges and sources of pollinators in a city, due to high abundance, richness, and diversity of pollinating insects. For our first objective, we hypothesize that flower visitor diversity will be higher in the natural environment than in urban environment. This is due to the predominance of exotic ornamental plants in urbanized areas. For our second goal, if visitor diversity instead responds to resource abundance and not to its status (native versus exotic), the higher density of flowers found in residential urban environments can contribute to increased visitor diversity. However, this effect will vary between natural and urban landscapes due to the higher diversity of exotic plants in urban environments. Therefore, we predict that the density of flowers will concentrate a greater diversity of visitors, but it will vary according to the proportion of exotic or native plants. Finally, for our third goal, we hypothesize that because within cities, green spaces provide various resources for floral visitors, they may act as diversity hubs; accordingly, we predict that the diversity of visitors will be greater in green spaces and it will decay as we get further away from parks and squares.

## Materials and Methods

### *Study area*

This study was developed in the Yungas ecoregion, a subtropical montane forest in the northwest of Argentina. The climate is subtropical with monsoon rains (dry winters and rainy summers); precipitation and temperature are controlled strongly by the topography (Grau et al. 2008). The piedmont of this forest has suffered the greatest human pressure, mainly due to its accessibility and the abundance of natural resources (Brown 2009). As a result, agriculture is developed in this landscape with growing urbanizations surrounded by some portions of natural forest. This has led to the development of the main urban centers of northwestern Argentina, thus promoting the degradation of the native vegetation and transformation of the environment (Gutiérrez Angonese 2015). This study took place in the city of Yerba Buena (26°37′ S - 65°41′ W; 500 m a. s. l.), located in the province of Tucumán. The city of Yerba Buena is predominantly a residential city with several commercial centers, public services and public transportation options, with a population of 102741 inhabitants according to the National Census of Population, Households, and Housing (INDEC 2022). This city extends to the west of the urban area of Gran San Miguel de Tucumán and at the piedmont of the SSJ Park on its eastern slope.

The SSJ Park is a protected natural area (14000 ha) belonging to the National University of Tucumán. Within the park, the abandonment of agricultural practices has allowed the development of secondary forests dominated in certain areas by exotic species such as privet (*Ligustrum lucidum*), blackberry (*Morus nigra*) and black acacia (*Gleditsia triacanthos*) (Grau et al. 2008). Present-day dynamics of land use in the urban agglomeration of Yerba Buena are characterized by a decrease in agricultural activities and an increase in urbanization associated with population growth (Oltra-Carrió et al. 2010).

### *Sampling*

Fieldwork was completed during the peak flowering season between September and December 2018, between 09:00 and 13:00.

We sampled during these months as they represent the period in which we encounter the highest abundance and diversity of concurrently flowering plants during the flowering season. We alternated urban sites and natural sites, one of each once a week. Days with temperatures below 18 °C or above 42 °C were excluded from the study, as these extreme conditions may impact floral visitors. Additionally, sampling did not occur on windy days.

Sampling was done in five squares located in the city of Yerba Buena, and five natural sites located in the Sierra San Javier Park, each of which was sampled only once. The squares selected were: Round Square (26°49'11.35'' S - 65°16'40.13'' W), Rubén Darío Square (26°49'10.17'' S - 65°16'58.40'' W), Nougués Square (26°48'58.76'' S - 65°17'33.06'' W), Moreno Square (26°48'25.29'' S - 65°17'36.01'' W) and Percy Hill Park (26°48'36.20'' S - 65°16'59.43'' W). With respect to the natural areas located in the SSJ Provincial Park, two sites were located adjacent to an isolated university-owned building (26°46'36.83'' S - 65°19'52.56'' W; 26°46'20.67'' S - 65°20'7.46'' W), and three were next to a hiking trail (26°47'47.52'' S - 65°19'46.83'' W; 26°48'9.04'' S - 65°19'31.14'' W; 26°47'41.31'' S - 65°20'9.47'' W). Sites from different environments were separated by 0.7 to 2 km (Figure 1). Within a single environment, each replicate was separated by a minimum of 500 m, well beyond the average distance of successful pollen transfer between insect-pollinated plant species in urban habitats (130 m) (Van Rossum 2010). While it is well known that several bees, particularly honeybees and bumblebees, can cover far greater distances within single foraging bouts, they have been shown to prefer to forage locally within the range of 500 m from nests in suburban and rural contexts (Garbuzov et al. 2015; Osborne et al. 2008).

In each site (five natural and five urban), we delimited four 500-m-long transects, along which we recorded flowers and their insect visitors. In urban sites, we placed the transects along each street surrounding the square. The sidewalk of the squares was considered as point 0 m, and the 500 m transect started in each corner (Figure 2). In the natural sites, the transects were traced from a random sampling point, following the four cardinal points in such a way that a similar sampling pattern was repeated in both the urban and the natural environments.

Along each transect, we defined sampling points every 20 m which consisted of a circle of 5 m diameter. In this area, we estimated the abundance of flowers. The number of

**Figure 1.** Satellite image of Yerba Buena City and east slope of Park SSJ.

**Figura 1.** Imagen satelital de la ciudad de Yerba Buena y la pendiente este del Parque SSJ.

**Figure 2.** Scheme of the way in which the transects were drawn and the censuses were carried out along them. The green lines surrounding the square correspond to the sidewalks within the square that were sampled. The red points at each corner correspond to the 0 m mark from where the 500 m transect (yellow line) begins.

**Figura 2.** Esquema de cómo se trazaron las transectas y se realizaron los censos a lo largo de ellas. Las líneas verdes que rodean a la plaza corresponden a las veredas dentro de la plaza que fueron muestreadas. Los puntos rojos en cada esquina corresponden al 0 m desde donde comienza a contarse la transecta de 500 m (línea amarilla).

flowers on each plant was estimated by direct counting of all open floral units. A 'floral unit' was defined as a separate structure where an insect lands to obtain nectar or pollen (Weiner et al. 2014) (i.e., in Onagraceae, the floral unit is composed of a flower, while in Asteraceae, the floral unit is an inflorescence). The abundance of open flowers at the time of sampling was calculated using categories according to the number of flowers on each plant (1, 10, 50, 100, 500, +1000), and the resource density was calculated for each sampling point as the number of floral units per sampling point area (Potter and LeBuhn 2015). We included plants whose height was lower than 2 m. In the case of herbaceous or creeping plants, we considered those that had a coverage of more than 0.5 m$^2$. Grasses were excluded from the sampling. Photographic records of all the selected plants along the transects were taken for their subsequent identification, for which we used the online catalog of Flora Argentina y del Cono Sur (floraargentina.edu.ar) and the Guía de San Javier (Grau 2021). We used these catalogs to classify plants as native or exotic to calculate the abundance of plants of each status in each sampling point.

In each area, we also recorded flower visitors during a 5-minute observation period (hereafter, census) focusing either on a branch of a focal plant, or the whole plant, depending on its size and/or accessibility. During each census, for each visiting insect we recorded the number of flowers visited, the number of flowers that were observed at each census, and the total number of flowers per focal plant. The insects that visited the observed flowers and came into direct contact with the fertile pieces of the plant's flower were considered potential pollinators and recorded as a visit. Individual insects were collected using entomological aspirators and/or entomological nets for identification through the use of taxonomic keys (Abrahamovich et al. 2007; Álvarez 2015; Álvarez et al. 2016; Brothers et al. 2016; Buck 2009; Chalup 2021; Coscarón and Carpintero 2023; Dalmazzo et al. 2020; Durante et al. 2008; Fernandez and Palacio 2006; González and Correjo 1992; Lucia et al. 2014; McAlpin 1981; Michener 2007; Roig-Alsina 2008, 2014; Ruz et al. 2008; Sarmiento 2006; Urban 2009; Werenkraut et al. 2022). The specimens that could not be identified to the species level were classified as morphospecies. All the material collected was deposited in the Entomological Collection of the Institute of Regional Ecology (CONICET-UNT).

### *Data analyses*

For our first goal, we quantified for each site in both natural and urban landscapes the abundance of visitors per flower, species richness and diversity measured as Shannon index (H′). The abundance was calculated as the number of visiting insects divided by the number of observed flowers, so that the data used for analysis was the number of visitors/flower, which is comparable between censuses and among studies. The Shannon diversity index is a widely used indicator that considers species richness and evenness in the distribution of individuals among species. Since the abundance and presence of the honeybee, *Apis mellifera*, in all the censuses greatly exceeded that of the rest of the floral visitors, we considered that it could mask other patterns that may emerge from our data on native pollinators. Therefore, all indices were evaluated in two different ways: including the abundance of honeybees and excluding it. A comparative analysis of diversity measures of pollinators between the sites in both environments was conducted using t-tests for comparison between means. This allowed the evaluation of potential differences between the natural and urban environments in terms of abundance, richness, and diversity of pollinators.

To address our second aim, for each urban and natural site, we quantified the richness of flowering plant species, diversity of flowering plants (measured as the Shannon index [H′]), abundance of exotic/native plants, and the density of flowers in each sampling site in both environments. We performed a generalized linear model (GLM) to explain the relationship between flower visitors with the type of environment (natural/urban), the density of flowers, the richness of plant species, and the proportion of exotic plant species, all of them considered as explanatory variables in the models. Total plant abundance and diversity of plants were omitted from GLM analyses due to their significant correlation with flower density and plant species richness, respectively, to avoid multicollinearity. In our models, we considered the abundance of flower visitors, Chao's estimator and species diversity (H′) as response variables. The Chao1 index is a non-parametric estimator of species richness, based on the number of rare species in the sample, those that appear only once (singletons) and those that appear twice (doubletons) (Colwell and Coddington 1994). Because Chao1 uses abundance rather than incidence data, it is more appropriate for species richness estimates of mobile organisms, such as insects (Brose and Martínez 2004). For the model, we assumed a Gamma distribution of the error due to the nature of our response variables, which likely had higher dispersion than expected under a normal distribution. This distribution enables us to capture greater variability and flexibility in modeling positive variables, as opposed to the normal distribution, which assumes constant dispersion (Crawley 2007). Furthermore, we used a log link function to ensure that the predicted values remained positive (Crawley 2007). To mitigate the influence of outliers, we applied a logarithmic transformation to flower density. This transformation helped to stabilize the variability of the variable, making it more suitable for statistical modeling and reducing the influence of extreme values on the results. The models were built with the glm function of the stats package in R Statistical Software, version 4.0.2. The parameters were estimated using the maximum likelihood method. In turn, we selected the best model with the step function, also from the stats package, considering the Akaike selection criterion and deviance (Crawley 2007). The function operated iteratively, adding or removing predictor variables based on a predefined criterion. It started by including all independent variables in the model and subsequently removed those that did not meet the established selection criterion (Crawley 2007). The model with the lowest AIC, and smaller deviance was considered the best. The selected model provides the optimal combination of fit and parsimony, making it the most suitable for explaining the relationship between the predictor variables and the target variable.

For our third goal, understanding whether urban green spaces could act as refuge and source of flower visitors in a city, we quantified abundance and diversity of visitors at squares and parks, and looked at whether those variables decreased with increasing distance from the urban green space. We calculated the abundance, Chao's estimator richness, and diversity of visitors for each sampling point across transects, and we used these as response variables along the distance gradient away from green spaces. We built generalized linear models (GLM) assuming local variables at each point the distance to the green space, the density of flowers, richness of plant species, proportion of exotic plants species, and diversity of plants. The modeling was carried out with the glm function of the stats version 4.0.2 package in R. The Gamma distribution error was used for modeling, like the previous models.

## Results

We carried out a total of 221 censuses, each one consisting of a 5-minute observation of a flowering plant, which corresponded to 1105 minutes of sampling effort. We recorded flower visitors over a total of 67 species of flowering plants belonging to 34 families (Supplementary Material 1-Table S1). Fabaceae was the most abundant plant family (10.15%), followed by Asteraceae (8.70%) and Solanaceae (8.70%). We recorded a total of 41 flowering plant species in urban areas and 26 in natural sites. Plant flowering species richness differed significantly between the landscapes (t=-2.31, df=8, P=0.05). Mean plant richness was higher in urban areas (12.60±4.40) compared to natural areas (8.00±0.70) per sample site. Plant flowering species diversity (H') also differed significantly, with higher average diversity in urban environments (H'=2.19±0.38) than in natural environments (H'=1.45±0.38) (t=-2.90, df=8, P=0.02) (Figure 3). These findings indicate that both plant flowering species richness and plant species diversity are significantly higher in the urban landscape compared to the natural landscape. Also, the proportion of exotic flowering plants was higher in the urban than in the natural landscape. In urban areas they represented 75% of plant species, while in natural areas they represented 20% (t=-12.00, df=8, P<0.05) (Figure 4).

We recorded 89 species of floral visitors, with Hymenoptera being the predominant order in both the natural and urban landscapes (Supplementary Material 2-Table S2). In the natural landscape, Hymenoptera constituted 59% of the individuals, followed by Diptera (16%) and Lepidoptera (14%). The most abundant families within Hymenoptera were Apidae (74%) and Halictidae (16%). In the urban landscape, Hymenoptera represented 85% of the recorded individuals, with Apidae representing 80% and Halictidae 6% of the species in the order. The orders Diptera, Lepidoptera and Coleoptera had relatively similar abundances, with nearly 5% each. Lepidoptera and Diptera showed lower abundances in the urban environment than the natural ones. Concerning the honeybee, *Apis mellifera* —which represents the most abundant species in both environments—, we obtained a total of 245 recorded individuals. In the natural environment, its abundance was slightly higher (n=135) than that of the urban environment (n=110) (Supplementary Material 2-Table S2).

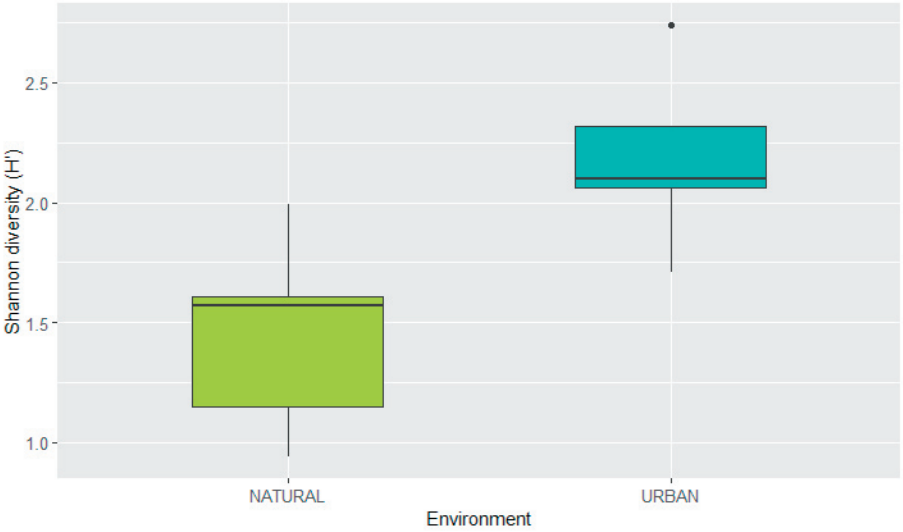

**Figure 3.** Boxplot graph displays the comparison of plant diversity between sites from two environments: natural and urban. The graphs used the environment as a factor (x-axis) and the respective diversity as a continuous variable (y-axis).

**Figura 3.** El gráfico de diagrama de caja muestra la comparación de la diversidad de plantas entre sitios de dos entornos: natural y urbano. Los gráficos utilizaron el ambiente como factor (eje x) y la diversidad respectiva como variable continua (eje y).

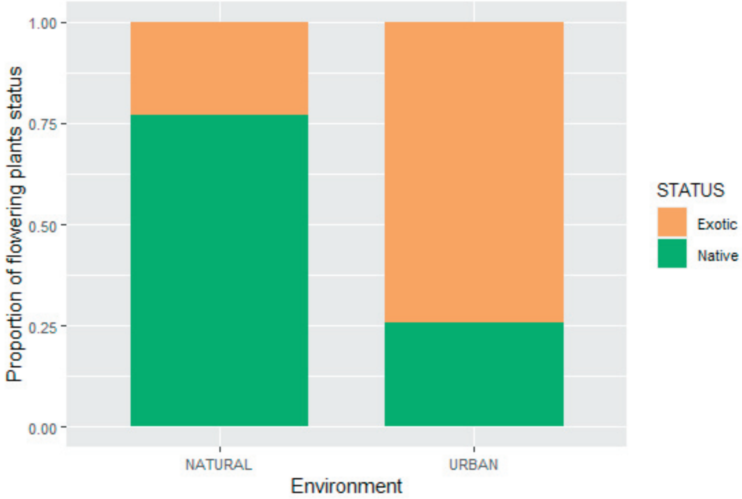

**Figure 4.** Proportion of exotic and native flowering plant species surveyed in an urban and a natural Yungas environment in Tucumán, Argentina. In natural areas, of the 97 specimens of flowering plant specimens recorded, seven were exotic (belonging to six species). The other 90 specimens were native and belonged to 20 species. In the urban environment, a total of 122 specimens of flowering plants were recorded, of which 20 belonged to 11 native species and 102 specimens to 32 exotic species.

**Figura 4.** Proporción de especies de plantas con flores exóticas y nativas registradas en un entorno urbano y natural de las Yungas en Tucumán, Argentina. En áreas naturales, de los 97 especímenes de plantas con flores registrados, siete eran exóticos (pertenecientes a seis especies). Los otros 90 especímenes eran nativos y pertenecían a 20 especies. En el entorno urbano, se registraron un total de 122 especímenes de plantas con flores, de los cuales 20 pertenecían a 11 especies nativas y 102 especímenes a 32 especies exóticas.

We found no significant difference in visitor abundance between the natural (3.16±1.96) and the urban environment (4.30±1.68) (t=-0.97, df=8, P=0.36). When excluding the abundance of *A. mellifera* from the analysis, results did not change: no significant differences were found between the mean abundance of visitors in the natural landscape (2.50±1.70) compared to the urban landscape (2.90±1.74) (t=-0.39, df=8, P=0.70). Species richness of flower visitors was not statistically different between natural and urban sites. However, it tended to be higher in the natural environment (15.8±5.12) compared to the urban environment (10.6±2.88) (t=1.9, df=8, P=0.09). As for floral visitor diversity, no significant differences were found between the natural (H'=2.10±0.52) and the urban environment (H'=1.63±0.40) (t=1.5, df=8, P=0.18), but when the abundance of *Apis mellifera* was not considered, diversity in the natural landscape (H'=2.25±0.37) became significantly higher than in the urban landscape (H'=1.69±0.39) (t=2.3, df=8, P=0.05) (Figure 5).

Results from the GLM analyses indicate that the significant predictor variables were plant richness and flower density, no matter whether we included or not *A. mellifera* in the analysis. In both cases, plant richness exhibited a positive effect, while flower density showed a negative effect on floral visitor abundance. Moreover, we found that, as the proportion of exotic plants increases, the estimation of visitor species richness, calculated through the Chao estimator, tends to decrease (Table 1).

The diversity of floral visitors was explained by urbanization, plant species richness and flower density. The negative effect of urbanization suggests that, compared to natural environments, the diversity of floral visitor species tends to decrease in urban settings. Similarly, we observed that an increase in flower density is associated with a decrease in floral visitor diversity (Figure 6). On the other hand, when analyzing the diversity of visitors without considering the abundance of *A. mellifera*, we found that only the proportion of exotic plants and flower density significantly explained it. Our results indicate that the presence of exotic plants has a significant negative effect on the diversity of non-honeybee visitors (Figure 7). On the other hand, flower density, although included in the best-selected model, exhibited statistically non-significant behavior. Nevertheless, it was observed to maintain a tendency to negatively impact floral visitor diversity (Table 1).

For urban environments, we constructed models to investigate whether flower visitors were more abundant in green spaces and decreased as we got further away. Due

**Figure 5.** Shannon diversity of flower visitors in both environments. The graphs used the environment as a factor (x-axis) and the respective diversity as a continuous variable (y-axis).

**Figura 5.** Diversidad de Shannon de visitantes florales en ambos ambientes. Los gráficos utilizaron el ambiente como factor (eje x) y la diversidad respectiva como variable continua (eje y).

**Table 1.** Table of the best models selected using the 'step' function for the abundance, richness, and diversity of floral visitors in two environments (natural/urban). Models were chosen based on the lowest AIC and the smallest deviance for optimal fit. The selected model provides the optimal combination of fit and parsimony, making it the most suitable for explaining the relationship between predictor variables and the target variable. The coefficients represent the estimated impact of each predictor variable accompanied by their respective standard errors, t-values, and p-values.

**Tabla 1.** Tabla de los mejores modelos seleccionados utilizando la función 'step' para la abundancia, la riqueza y la diversidad de los visitantes florales en dos entornos (natural/urbano). Los modelos se eligieron en base al valor de AIC más bajo y la deviance más pequeña para su ajuste óptimo. El modelo seleccionado proporciona la combinación óptima de ajuste y parsimonia, por lo que es la más adecuada para explicar la relación entre las variables predictoras y la variable respuesta. Los coeficientes representan el impacto estimado de cada variable predictora es acompañada de sus respectivos errores estándar, el estadístico t y valores p.

| Best model selected | term | estimate | std.error | statistic | p.value |
|---|---|---|---|---|---|
| Abundance of Flower Visitor ~ Plant Richness + Log (Flower density) | (Intercept) | 6.18 | 1.17 | 5.29 | <0.01 |
| | Plant Richness | 0.15 | 0.04 | 3.65 | <0.01 |
| | Log (Flower density) | -1.10 | 0.22 | -4.94 | <0.01 |
| Abundance of Flower Visitor without A. mellifera ~ Plant Richeness + Log (Flower density) | (Intercept) | 6.81 | 1.91 | 3.57 | <0.01 |
| | Plant Richness | 0.15 | 0.07 | 2.28 | 0.05 |
| | Log (Flower density) | -1.24 | 0.35 | 0.35 | 0.01 |
| Richness Chao's estimator ~ Proportion of Exotic Plants | (Intercept) | 3.73 | 0.19 | 18.96 | <0.01 |
| | Proportion of Exotic Plants | -4.55 | 1.52 | -2.99 | 0.02 |
| Diversity of flower visitors ~ Landscape (Urban) + Plant Richness + Log (Flower density) | (Intercept) | 2.39 | 0.51 | 4.73 | <0.01 |
| | Landscape (Urban) | -0.50 | 0.14 | -3.36 | 0.01 |
| | Plant Richness | 0.06 | 0.02 | 2.59 | 0.04 |
| | Log (Flower density) | -0.36 | 0.10 | -3.63 | 0.01 |
| Diversity of flower visitors without A.mellifera ~ Proportion of Exotic Plants + log(Flower.density) | (Intercept) | 1.53 | 0.52 | 2.94 | 0.02 |
| | Proportion of Exotic Plants | -0.39 | 0.15 | -2.64 | 0.03 |
| | Log (Flower density) | -0.12 | 0.08 | -1.38 | 0.21 |

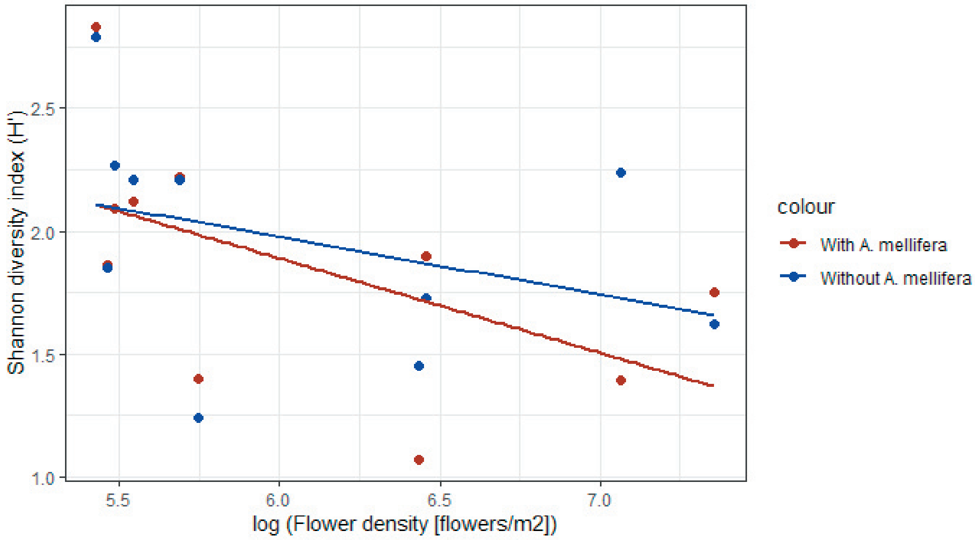

**Figure 6.** Influence of flower density on the diversity of flower visitors with (red dots and line) and without (blue dots and line) *A. mellifera*.

**Figura 6.** Influencia de la densidad de flores en la diversidad de visitantes florales con (puntos y línea roja) y sin (puntos y línea azul) *A. mellifera*.

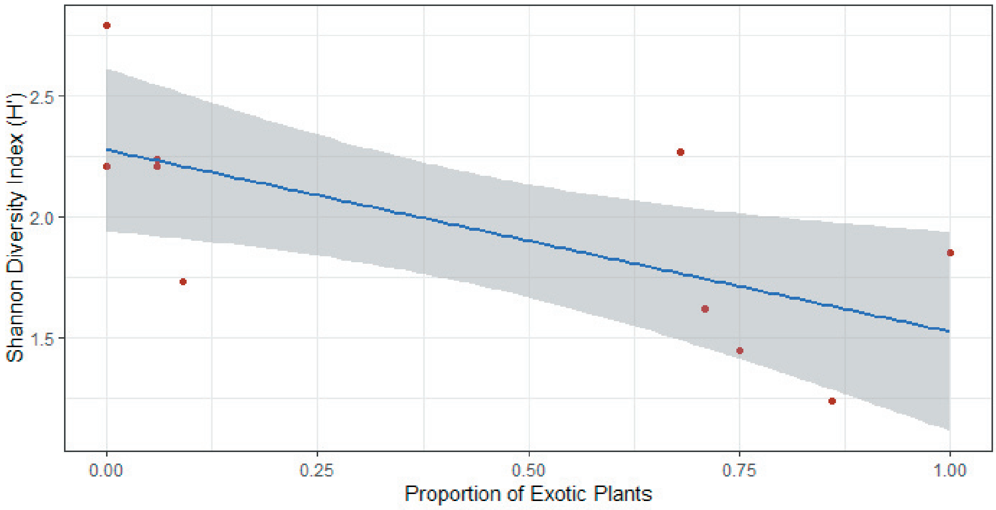

**Figure 7.** Influence of the proportion of exotic plants on the diversity of flower visitors without *A. mellifera*.

**Figura 7.** Influencia de la proporción de plantas exóticas en la diversidad de visitantes florales sin *A. mellifera*.

to the identified correlation between our explanatory variables (distance and plant richness), we included the interaction distance: plant richness in our models. This interaction aims to explore how the relationship between distance and the response variable may vary based on the levels of plant richness in the environment (Table 2).

In the best models selected for the abundance of floral visitors, considering or not considering the abundance of *A. mellifera*, our analyses revealed that no predictor affected this variable (Table 2). On the other hand, distance to green spaces and the proportion of exotic plants in the sampling point were the best predictors for species richness, as estimated by the Chao index (Table 2).

In terms of diversity, our two models, with and without *A. mellifera*, behaved similarly (Table 2). The variables that best explained pollinator diversity were distance to green spaces and plant richness. Again, distance

**Table 2.** Table of the best models selected using the 'step' function for the abundance, richness, and diversity of pollinators at increasing distances from green spaces in an urban landscape. Models were chosen based on the lowest AIC and the smallest deviance for optimal fit. We included the interaction "distance: plant richness" in our models. This interaction aims to explore how the relationship between distance and the response variable may vary based on the levels of plant richness in the environment. The coefficients represent the estimated impact of each predictor variable accompanied by their respective standard errors, t-values, and p-values.

**Tabla 2.** Tabla de los mejores modelos seleccionados utilizando la función 'step' para la abundancia, riqueza y diversidad de polinizadores a distancias crecientes desde espacios verdes en un entorno urbano. Los modelos fueron elegidos en función del AIC más bajo y la desviación más pequeña para un ajuste óptimo. Incluimos la interacción "distancia: riqueza de plantas" en nuestros modelos. Esta interacción tiene como objetivo explorar cómo la relación entre la distancia y la variable de respuesta puede variar según los niveles de riqueza de plantas en el entorno. Los coeficientes representan el impacto estimado de cada variable predictora acompañado de sus respectivos errores estándar, estadístico t y valores p.

| Best model selected | term | std.error | statistic | p.value |
|---|---|---|---|---|
| Flower Visitor Abundance ~ Distance + Proportion of Exotic Plants + Plant Richness + Distance: PlantRichness | (Intercept) | 0.16 | -0.08 | 0.93 |
| | Distance | 0.01 | 0.91 | 0.36 |
| | Proportion of Exotic Plants | 0.11 | 1.47 | 0.14 |
| | Plant Richness | 0.07 | 1.52 | 0.13 |
| | Distance:PlantRichness | 0.01 | -1.42 | 0.16 |
| Chao visitors ~ Distance + Proportion of Exotic Plant | (Intercept) | 0.95 | 4.39 | < 0.01 |
| | Distance | < 0.01 | 2.23 | 0.03 |
| | Proportion of Exotic Plant | 0.21 | -1.77 | 0.08 |
| Diversity of flower visitors ~ Distance + Plant Richness | (Intercept) | 0.10 | 2.28 | 0.02 |
| | Distance | < 0.01 | -2.28 | 0.02 |
| | Plant Richeness | 0.05 | 2.22 | 0.03 |
| Diversity without A.mellifera ~ Distance + Plant Richness | (Intercept) | 0.09 | 1.62 | 0.12 |
| | Distance | < 0.01 | -2.51 | 0.01 |
| | Plant Richeness | 0.05 | 2.22 | 0.03 |

negatively affected floral visitors, although its effect was very low if short distances to green spaces are considered. However, the richness of plant species positively affected the diversity of floral visitors.

## Discussion

Urbanization alters the composition of biotic communities by creating a matrix of habitats distinct from natural ecosystems (McKinney 2008). In this study, where we investigated the composition of urban communities of floral visitors with respect to the floral resources provided by the urban environment (e.g., nectar, pollen, nesting sites, etc.), we observed a tendency to decrease both in richness and diversity of floral visitors in the city compared to natural sites nearby. This is consistent with our first prediction, which postulates that the diversity in urban landscapes will be lower than in natural landscapes. Also, the richness of exotic flowering plants negatively affected the abundance and richness of flower visitors.

Although the diversity of flowering plants in the urban environment was greater than in natural habitats, many cultivated plants that enrich the urban space are not visited by insects (Lowenstein et al. 2019). In our system, this greater richness and diversity of flowering plants found in the city was not associated at the landscape level with an increase in the diversity of floral visitor communities. However, at the local level, increases in plant richness were associated with a higher abundance and diversity of floral visitors at all sites. Since the paired sampling between natural and urban sites was spaced out by one-week, potential weather events, such as storms, may have affected the number of flowers available. However, the units of comparison used for abundance and diversity calculations were floral visitors per flower, which allowed us to control for such potential variation when comparing diversity between the two landscapes.

Regarding the composition of the plant community according to its exotic/native status, both native and exotic plants serve as important sources of food for urban pollinators (Baldock et al. 2019). In our study system, exotic plants were associated with a decrease in pollinator diversity as predicted according to our second hypothesis. Although exotic plants play an important role as a food source when native resources are scarce (Baldock et al. 2019; Zaninotto et al. 2023), their presence can have a negative impact on pollinator species richness and diversity. This is because these exotic plants can provide resources or floral characteristics that could make them less attractive or compatible with native pollinators (Zaninotto et al. 2023). Most bees found in urban areas are known to be generalists and exploit a wide range of food resources (Fitch et al. 2019). Because we did not assess the diet of floral visitors, our findings do not demonstrate that flower visitors do not prefer exotic plants. However, in our system, the decrease in visitor diversity may have been influenced by an increase in the number of exotic plants. In fact, the most abundant floral visitor turned out to be an exotic generalist species, *A. mellifera*, with the ability to make use of a wide diversity of plants as foraging resources. However, the integration of native and exotic floral resources could be a strategy to promote the diversity and abundance of other floral visitors in urban environments (Persson et al. 2022b). Considering that, in our system, the proportion of exotic plants was higher in the urban landscape and the abundance of *A. mellifera* has been found to negatively impact the richness of wild bee species (MacInnis et al. 2023), an alternative explanation could be that the lower diversity found in urban spaces is the result of a synergistic effect of these two factors. Furthermore, it is important to consider that certain urban stressors, such as heat, pollution, and habitat fragmentation, can interact to create more adverse environmental conditions for pollinators. These factors affect biotic interactions, including foraging behavior and food choices (Buchholz and Kowarik 2019).

Concerning flower density, our models suggest a decrease in the diversity and abundance of floral visitors as flower density increases. This phenomenon could result from a dilution effect on pollinators within the floral neighborhood (Wenninger et al. 2016). Social bees, due to their ability to communicate the location of foraging resources to other workers, tend to increase the relative abundance and frequency of visitation in a focal plant. In contrast, solitary bees spend more time foraging in a dispersed manner (Conner and Neumeier 1995; Wenninger et al. 2016). However, it is important to consider that the relationship between flower density and floral visitors also depends on the sampling unit used to measure flower density (Totland and Matthews 1998). Taking this into account, it is possible that our sampling units were relatively small and it is likely that floral visitors also simultaneously visiting other nearby flower neighborhoods simultaneously, either due to floral display or the quality and quantity of available resources in those locations.

According to our third prediction, we expected a decrease in pollinator diversity as the distance to green spaces increased. As reported by other authors, greater diversity of floral visitors is associated with proximity to green areas within the urban landscape (Ahrné et al. 2009; Dylewski et al. 2019; Glaum et al. 2017; McFrederick and LeBuhn 2006; McIntyre and Hostetler 2001). According to our results, this relationship is evident when analyzing the diversity of flower visitors. The effect of distance to green spaces, although quite slight, becomes relevant at distances greater than 300-500 meters. These distances are considered the foraging ranges of many species of solitary bees and certain colonial bees (Garbuzov et al. 2015; Osborne et al. 2008). However, this effect may also have been influenced by the diversity of private backyards adjacent to the transects. However, our findings suggest that flower visitor communities are more significantly influenced by other variables of interest, such as the richness of plant species and the proportion of exotic plants. We should consider that green spaces in our study system may not represent the sole source of food and nesting resources for urban floral visitors (Davis et al. 2017; Fischer et al. 2016; McFrederick and LeBuhn 2006). Some of the key factors that also influence urban flower visitors' communities in neighboring gardens were not studied in our system, such as the surface of permeable soil available for nesting and the proportion of flowering plant species for nectar, both native and exotic (Daniels et al. 2020; Threlfall et al. 2015). These factors appear to play a crucial role in shaping pollinator communities and will therefore impact organisms that provide pollination services in urban sites.

The composition of the plant community in urban environments, determined by management practices, urban planning policies, and the degree of urbanization, influences the composition of pollinator communities that persist in cities (Dylewski et al. 2020). Urban sites with limited management tending to preserve a certain similarity of their natural structure are habitats that provide high-quality environments for the development of pollinator communities (Dylewski et al. 2019, 2020; Kearns and Oliveras 2009). Although we do not differentiate the degree of management between the different green spaces in our system, we observed that one area —in particular, Percy Hill Park— had little pruning and high vegetation cover, as it is considered a relict of Yungas immersed in the city of Yerba Buena. This green space concentrated the greatest diversity and abundance of pollinators in the city, which is consistent with previous findings in green spaces with little or no management within cities (Dylewski et al. 2019, 2020; Kearns and Oliveras 2009). Therefore, anthropic pressure in urban areas can interrupt different biotic processes, subjecting the biological communities to constant changes to adapt to disturbances and alterations imposed by human activities (Sattler et al. 2010), yet certain management practices can help reduce the negative impact of such changes.

In conclusion, we have observed that floral visitor communities in urban environments are less diverse than those at natural sites, primarily due to the type of vegetation available for foraging. Urbanization with low population density and residential areas with open green spaces have been shown to have a positive effect on pollinator foraging and nesting (Wenzel et al. 2020). The diversity of plant species recorded in the city, mainly exotic, is a clear reflection of the structure of the city and the preference of its human inhabitants, and of how the presence of this exotic plant community, among other factors, may condition urban floral visitors. We find it appropriate to suggest that cities without intensive urbanization, and that still share many species with the closest natural space, may focus on urban planning that considers development alternatives that contribute to the conservation of potential pollinators.

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