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The impact of impervious surface and neighborhood wealth on arthropod biodiversity and ecosystem services in community gardens

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Abstract

As the number of humans living in cities has grown, interest in the value of community gardens to provide agricultural products has increased. However, neighborhoods with different land cover patterns and socioeconomic characteristics often differ in their ecological attributes, leading to potential differences in biodiversity-mediated ecosystem services (e.g., pollination and pest control). Here we ask, how do impervious surface and socioeconomic features of the urban matrix around community gardens impact arthropod biodiversity and pollination and pest control services? We collected arthropods (insects, arachnids, myriapods, and isopods) across community gardens in Boulder Co., CO, and used experimental jalapeño pepper plants as a sentinel crop to measure herbivory damage and pollination services. We categorized arthropods into functional guilds to see how impervious surface and neighborhood wealth in the urban matrix surrounding a site impacts the abundance of three focal groups – pollinators, herbivorous pests, and predators. We also looked at how bee Hill-Simpson diversity responded to these variables. Through structural equation modeling, we found that fruit size increased as bee biodiversity increased, and bee biodiversity and overall pollinator abundance were negatively related to neighborhood wealth. Additionally, pollinator abundance was lower in gardens surrounded by higher amounts of impervious surfaces. Neighborhood wealth and impervious surfaces were positively correlated with herbivore and predator abundances, but these abundances had no relationship with herbivory damage in our plants. This research shows that reducing the amounts of impervious surface in the urban matrix can help increase bee biodiversity and abundance and improve pollination services in urban community gardens.

Keywords Community gardens · Arthropod-mediated ecosystem services · Urban matrix · Pollinators

Introduction

Community gardens serve multiple ecological and societal roles in cities (Pearson and Hodgkin 2010; Cabral et al. 2017; Clarke et al. 2019). They provide agricultural products that may be especially valuable in low-income areas deemed "food deserts," where residents have limited access to fresh fruits and vegetables (Krishnan et al. 2016). These gardens also serve as habitat patches for arthropods (Clucas et al. 2018; Baldock et al. 2019), which then impact food production through pollination, herbivory, pest control, and soil nutrient cycling services (Altieri 1999; Zhang et al. 2007; Lin et al. 2015). Maintaining diverse arthropod communities may be necessary for reliable pollination (Iuliano et al. 2017) and pest control ecosystem services (Bianchi et al. 2006) for an increasingly urban human population.

Anthropogenic land use often has negative effects on arthropod biodiversity. For instance, highly urbanized and economically disadvantaged areas tend to have fewer and lower-quality floral resources and nesting sites for pollinators (Zhao et al. 2019; Ferrari and Polidori 2022). Cities are also covered with impervious surfaces, which reduce habitat for most arthropods and may produce a "heat island effect," which can strongly impact their ectothermic physiology (Hamblin et al. 2018; Fenoglio et al. 2021). Previous studies have also shown that urban greenspaces host fewer arthropod natural enemy predators and more herbivorous pests than non-urban natural areas (McIntyre 2000; Pickett et al. 2001; Korányi et al. 2022). Managing insect pests is a top concern among community garden site users (Gregory

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et al. 2015; Liere et al. 2020); therefore, maximizing the pest control and pollination services provided is particularly relevant to applied urban ecology research, which emphasizes improving human wellbeing (Pickett et al. 2016; Byrne 2022).

In addition, urban ecology research has increasingly emphasized that the urban matrix is a complex, heterogeneous landscape with feedback between human and natural components (Pickett et al. 2016). This heterogeneity in landscape patterns can be difficult to capture and thus explains why recent studies have yielded conflicting results on arthropods' responses to urbanization metrics (Theodorou et al. 2020; Wenzel et al. 2019). For example, Matteson et al. (2012) found that highly developed urban areas with dense infrastructure had lower bee biodiversity, while Lowenstein et al. (2014) saw the opposite trend. Some of this variation may be due to factors like impervious surfaces and urban development (both derived from land cover data and often serving as a metric for urbanization) being insufficient predictors of matrix quality. Impervious surface measurements can conceal land use heterogeneity and complex vegetation within smaller parcels of land in urban areas (Cadenasso et al. 2007; Beninde et al. 2015). To better predict matrix quality, scientists have started incorporating social variables with impervious surfaces into landscape-level urban ecology research (Pickett et al. 2016; Zhao et al. 2019).

Studies have found that urban biodiversity positively correlates with socioeconomic conditions (i.e., income, home values, and educational attainment; Hope et al. 2003; Leong et al. 2018). This pattern is called *the luxury effect*. One study to uncover this relationship in arthropod communities found that neighborhoods with higher income had higher indoor arthropod diversity and that this relationship was stronger in homes with less yard space (Leong et al. 2016). This supports past research findings that cities' larger-scale landscape and economic factors may impact biodiversity patterns more than local habitat factors (Goddard et al. 2010; Mitchell et al. 2014). To date, few studies have attempted to link landscape-level land cover and economic covariates to ecosystem services as a function of arthropod community composition in urban agricultural systems. However, studies have identified links between different types of urban green spaces and arthropod-mediated ecosystem services, showing that urban landscape context does impact the quality of pest and pollination services (Gardiner et al. 2013; Bennett and Lovell 2019; Philpott and Bichier 2017).

The overarching question for this study is: How well do the social-ecological features of the urban matrix predict the pollination and pest control services provided by arthropods in community gardens? We hypothesize that both biophysical and socioeconomic factors surrounding garden sites will interact to shape arthropod communities and their ecosystem services by altering matrix quality. Specifically, we predict that increased impervious surfaces will reduce pollinator and natural predator abundances, as well as bee diversity, reducing pollination and increasing the abundance of herbivorous pests, which will reduce crop productivity. Conversely, increased neighborhood wealth will increase arthropod populations through improved matrix quality (increased habitat and floral resources), leading to an increase in pollination and pest control services, which will increase fruit mass (Fig. 1a). This work aims to understand the mechanisms that moderate arthropod communities and their services in urban gardens to better inform their management.

Methods

Study sites and Socio-ecological landscape context

The study was conducted at seven community garden sites in Boulder County, Colorado, USA (Fig. 1b). Boulder is a part of the Front Range Urban Corridor along the southeastern base of the Rocky Mountains with a semi-arid climate. The gardens in this study were all part of the same garden collective, Growing Gardens (https://growinggardens.org/), and ranged in size from 336 m^2 to 18,238 m^2 (mean 4110.8 ± SD 6008.0). Each garden was separated from the others by a minimum of 1.5 km and contained numerous plots individually maintained by members (Table 1). We do not know which specific fertilizers, herbicides, and insecticides were used in every plot, as this was at the discretion of individual gardeners. However, the Growing Gardens collective has a strict policy against synthetic products and only allows organic products. They also do not allow bee hives to be placed in the gardens. Vegetation was variable across individually managed plots, and we did not observe any systematic patterns that we expect would influence the results. Across the sites, some of the most common plants observed were squash (Cucurbitaceae), tomatoes and peppers (Solanaceae), and sunflowers (Asteraceae).

At the landscape scale, degree of urbanization was classified using the urban imperviousness landcover raster from the 2021 National Land Cover Database (NLCD, 30 m resolution) in R Statistical language (v4.2.0; R Core Team 2022) using the packages raster (v3.6.3; Hijmans R 2023), rgeos (v0.5.9; Bivand and Rundel 2021), rdgal (v1.5.32; Bivand et al. 2023), and sp (v1.5.0; Pebesma and Bivand 2005). Each pixel in the Urban Imperviousness raster shows impervious surfaces as a percentage of developed surfaces over 30 m^2. To find the most appropriate scale to measure impervious surfaces, we tested five different circular buffers with radii between 200 m and 2000 m around each

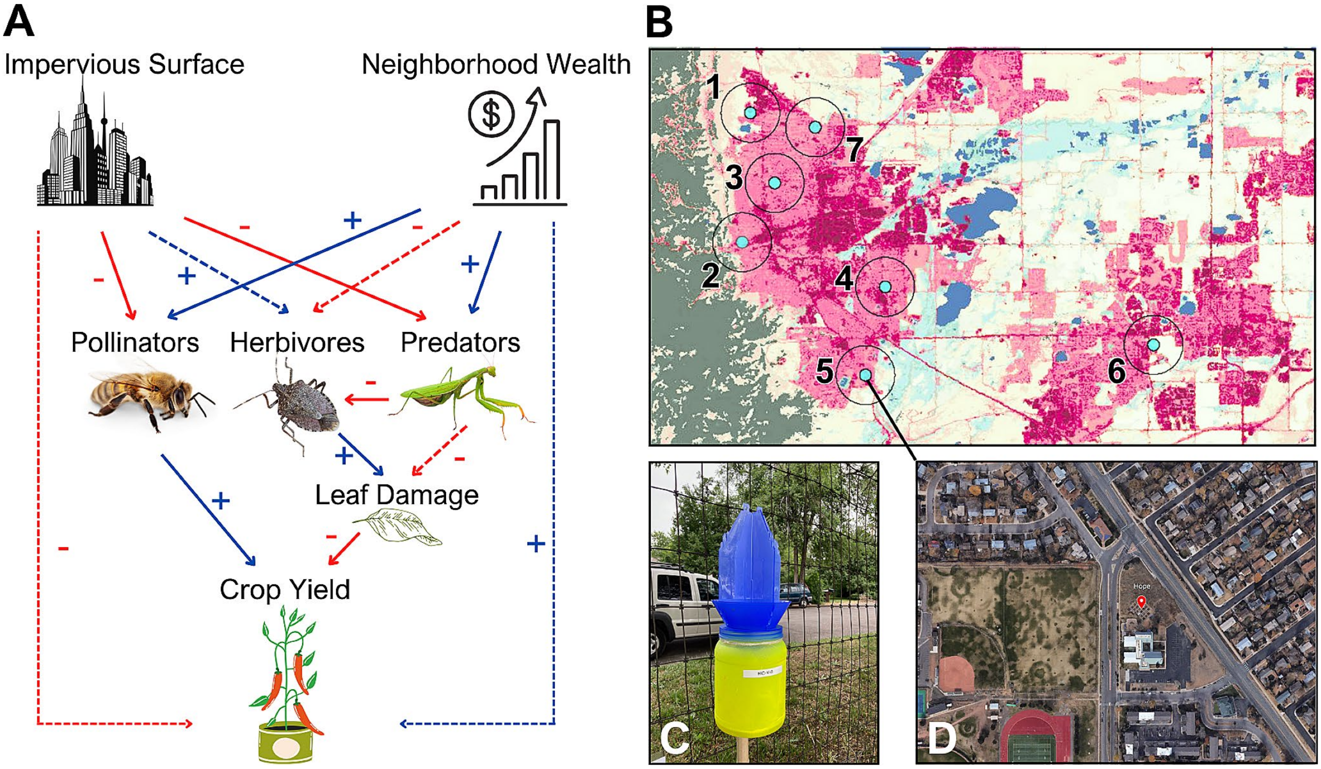

Fig. 1 (**A**) A conceptual diagram of predicted causal relationships between impervious surface and neighborhood wealth, arthropods, and crop yield. Red arrows indicate negative relationships, blue arrows indicate positive ones, solid lines indicate direct relationships, and dashed arrows indicate indirect ones. (**B**) A landcover class map of the study area, Boulder Co., CO, with community gardens (blue dots) and a 1 km buffer. Developed land cover is shown in magenta. Numbers correspond with garden names listed in Table 1. (**C**) A vane trap set out at a garden site. (**D**) An aerial photo of a community garden. Image credits: (**A**) Canva Pro License, (**B**) ESRI CC BY-NC, (**C**) Asia Kaiser, (**D**) Google Earth Images

Table 1 Number of plots, size, percent impervious surface within a 200-meter radius, Wealth Index values within a 1-km radius for each garden site in Boulder, CO, USA.

	Site	Number of plots	Garden size (m^2)	% Impervious surface (200 m radius)	ESRI Wealth Index (1-km radius)
1)	Foothills	46	3426.24	30.85	163
2)	Fortune	18	336.88	37.46	121
3)	Hawthorne	200	18238.58	22.68	235
4)	Hickory	28	2736.12	28.94	135
5)	Hope	23	657.99	36.42	159
6)	Kerr	41	2324.05	10.79	162
7)	Living Harvest	30	1055.85	33.29	209

garden site, then calculated the log-likelihoods of each model (models described below). The 200 m scale showed the best fit and was included in our final models (Fig. S1). The percent impervious surface values across our sites ranged from 19.06 to 34.34% (mean 28.6% ± SD 8.82%). We used ESRI's 2022 Wealth Index (ESRI Demographics, 2022) to quantify the economic conditions surrounding each garden site. In ESRI's wealth index, a value of 100 represents wealth equivalent to the national average, and it encapsulates household income, net worth, and the value of possessions and resources (Fig. S2). This Wealth Index was derived using ArcGIS software for residents within a 1-km radius of each community garden, the finest grain for which it was available for all gardens. The wealth index values across our sites ranged from 121 to 235 (mean 169 ± SD 37.9).

Arthropod sampling

We sampled arthropods in each garden using pitfall and vane traps on June 22–24, July 20–22, and August 19–21 of 2022. Three 13 cm deep white pitfall traps were placed at each site, equidistant from each other, with approximately 10 m of space between each trap. A funnel with a diameter of 3.5 cm was placed atop the traps to prevent vertebrate animals from falling in. The pitfall traps were placed in the ground immediately outside of plots in garden pathways. These are dirt paths where garden members regularly remove weeds and forbs. Three fluorescent blue vane traps with three different collection jar colors – clear, fluorescent yellow, and fluorescent magenta – were put in the field to help capture a greater diversity of flying insects (Vrdoljak and Samways 2011). The vane traps were placed at a height of 1.3 m (Fig. 1c) on stakes approximately 10 m apart from each other in the pathways in the gardens. Both pitfall and vane traps were filled with a propylene glycol-water solution to help capture and preserve specimens and left in the field for 48 h. The traps were then returned to the lab for specimens to be counted and identified to order level. Bees were further identified to species using taxon-specific books and dichotomous keys (Gibbs 2011; LeBuhn 2013; Wilson and Messinger 2016; Messinger Carril and Wilson 2023), Discover Life Identification Nature Guides (Ascher and Pickering 2017), and consultation with taxonomic experts from the University of Colorado Museum of Natural History (CUMNH) (Scott et al. 2011). Hemiptera and beetles were further identified to family or genus level, using Dunford and Long (2002), Discover Life Identification Nature Guides, and local expert opinion from the Resasco Lab at the University of Colorado Boulder. Lower-level classification helped us assign individuals more accurately to functional groups, as there is more variability in the dietary behavior in these clades. We were unable to accurately identify all small Hemiptera and beetle specimens (less than 5 mm in width), so their functional group was left unknown (Table S2). All arthropods were assigned to a functional group (pollinator, herbivore, predator, omnivore, detritivore, parasitoid, granivore, sanguinivore) based on their known feeding behavior (Schuh et al. 2010; Gullan and Cranston 2014; BugGuide, Iowa State University). Only pollinator, herbivore, and predator data were included in our final analyses.

Experimental plants

We used ‘Early Jalapeño’ cultivars of the species *Capsicum annuum* (Solanaceae) as sentinel plants to measure pollination services and herbivory damage because they grow well in various climates throughout the continental United States and benefit from insect pollination (Raw 2000). While jalapeños are self-pollinating and will set fruit with no pollination, fruit size is greater when they are pollinated (Cohen et al. 2020). They are also visited by a variety of insect pollinators, including bees (Raw 2000), wasps (Bosland and Votava, 1999), and syrphids (Jarlan et al. 1997). A total of 70 pepper plant seeds were planted in a greenhouse on April 1, 2022, and then placed in the gardens on August 17, 2022. At each site, five randomly chosen plants were covered loosely with mesh bags made of < 1 mm mesh tulle to prevent insects from landing on them, and five others were left exposed to examine the relationship between insect access and fruit mass and herbivory. The mesh coverings did not appear to inhibit the growth of the pepper plants (Asia Kaiser, personal observation). The plants were placed in the pathways at garden sites in two rows approximately 30 centimeters apart. They were left in the field for three weeks and watered every other day, then returned to the greenhouse. Any visible arthropods were removed from the plants, and all plants were covered with mesh upon returning them to the greenhouse to help prevent any unseen arthropods from spreading between plants from different sites. Any flower buds that grew in the greenhouse were pinched off to ensure we were only collecting data for peppers that had been pollinated in the field. When peppers began to turn red, they were picked and weighed fresh. The masses of all jalapeño peppers from a single plant were averaged to obtain a single fruit mass value per plant. Plant leaf damage, used as an index for herbivory, was calculated by counting the proportion of leaves on a plant with signs of herbivory damage (holes or chewed edges) promptly after plants were returned to the greenhouse.

Data analyses

Analyses were conducted using the R Statistical language (v4.2.0; R Core Team 2022) on macOS Big Sur 11.6.7. All data cleaning and data visualization was done using the Tidyverse package in R (Wickham et al. 2019).

Arthropod functional group analyses

We ran Negative Binomial Generalized Linear Mixed Models (GLMMs) with a log link to analyze the relationships between impervious surface and wealth index and total arthropod abundance, as well as the abundance of each functional group (pollinators, herbivores, and predators), for the full summer using the R package lme4 (v1.1.30; Bates et al. 2019). *Percent impervious surface* and *wealth index* were included as fixed effects and *month* was included as a random intercept in each model. We back-transformed the coefficient estimates provided by each model by taking the exponent of each coefficient to obtain effect sizes. To analyze the impact of impervious surface and wealth index on bee diversity (how we calculated diversity is described below), we ran a Linear Mixed Effect Model (LMM) with the same structure as the previous model with the R package nlme (v3.1.157; Pinheiro et al. 2022). We performed a natural log (ln) transformation for the bee diversity data to achieve a normal distribution. Additionally, we ran LMMs to test for differences in herbivory damage and pepper mass between our covered and uncovered pepper plants with *site* included as a random intercept. We then ran a Tukey test to look at pairwise differences between our sites.

Bee diversity

We examined bees in more detail using the Hill-Simpson Diversity Index with an asymptotic estimator to estimate relative bee diversity per site using the R package iNEXT (v2.0.2; Hsieh et al. 2016). This method has been demonstrated to provide a more appropriate metric for biodiversity than species richness, which is highly sensitive to rarity and can give more biased estimates of relative abundances at smaller sample sizes (Simpson 1949; Roswell et al. 2021).

We were also interested in how bee community composition varied across sites and whether or not this variation was linked to any environmental covariate. Bee community composition for each site was analyzed using all bees collected from vane and pitfall traps using Non-metric Multidimensional Scaling Analysis (NMDS) with Bray Curtis distances in the R package vegan (v2.6.4; Oksanen et al. 2022). This non-parametric indirect gradient analysis technique uses ranks rather than distances between objects and is, thus, more robust to data without an identifiable distribution (Kenkel and Orloci 1986). To analyze the effect of our environmental variables on bee community composition, we used the envfit function in the R package vegan (v2.6.4; Oksanen et al. 2022). With this function, we calculated a multiple regression of the environmental variables of interest, *percent impervious surface* (continuous) and *wealth index* (continuous), with the ordination axes from our NMDS. With this function, significance is tested by a permutation test.

Structural equation modeling

We constructed a structural equation model using the piecewiseSEM package in R (v2.3.0; Lefcheck 2015) to test the direct and indirect effects of our environmental covariates on herbivory damage and fruit sizes in our uncovered experimental jalapeño pepper plants. This model used only arthropod data from August to ensure that we were testing relationships with the arthropod community present at the same time our plants were there. Structural equation models are advantageous in that variables can be included as both predictors and response variables. Our structural equation model tested all the hypothesized relationships shown in Fig. 1a and consisted of six equations (Supplement 1) testing the following response variables: *bee Hill-Simpson diversity*, *pollinator abundance*, *herbivore abundance*, *predator abundance*, *leaf damage*, and *pepper mass*. *Wealth Index* and *impervious surface* were included only as predictor variables. Bee diversity, pollinator abundance, and herbivore abundance were encoded as correlated errors in the model. A directed-separation test in piecewiseSEM showed no significant dependencies between variables not tested against each other in the model.

Results

A total of 3287 arthropods were sampled across all community garden sites. Of this total, 1176 (34.5%) were pollinators, 368 (10.8%) were herbivores, and 255 (7.5%) were predators. 96% of pollinators were bees from 48 species across all gardens (Table S1). The most common taxa observed were long-horned bees (*Melissodes agilis*, $n = 413$; *Melissodes bimaculatus*, $n = 208$) and squash bees (*Eucera pruinosa*, $n = 296$). Most arthropods categorized as predators were arachnids ($n = 211$) and centipedes ($n = 30$). There were 24 different taxa categorized as herbivores (listed in Table S2); some of the most commonly collected herbivores were weevils (Curculionidae, $n = 29$) and grasshoppers (Acrididae, $n = 21$).

For our experimental jalapeño pepper plants, we found no significant difference in leaf damage between covered and uncovered plants (t = 0.432, df = 54.1, $p = 0.667$). There were also no differences in leaf damage across our sites ($p > 0.05$ for all pairwise differences). A total of 297 jalapeño peppers were harvested across 64 plants. Of the 70 plants initially placed in the field, 6 pepper plants (2 uncovered and 4 covered) never produced any fruit and were removed from our analyses. There were no significant differences in pepper mass between our covered and uncovered plants (t = -0.74, df = 54.5, $p = 0.46$).

Arthropod functional groups

There was no significant relationship between site percent impervious surface (z = -1.7, $p = 0.09$) or wealth index (z = -0.13, $p = 0.89$) and the average arthropod abundance per site for the summer (Fig. S3). However, we found significant relationships between our environmental covariates and pollinator, herbivore, and predator abundances (monthly data in Fig. S4). Pollinator abundance negatively correlated with impervious surface (z = -3.23, $p < 0.01$, Fig. 2a) but was unrelated to site wealth index (z = -1.48, $p = 0.14$, Fig. 2b). Every percent increase in impervious surface corresponded to a decrease in pollinator abundance by 0.96 (± 0.01 SE) individuals. Herbivore and predator abundance both positively correlated with impervious surfaces. Every percent increase in impervious surface corresponded to a

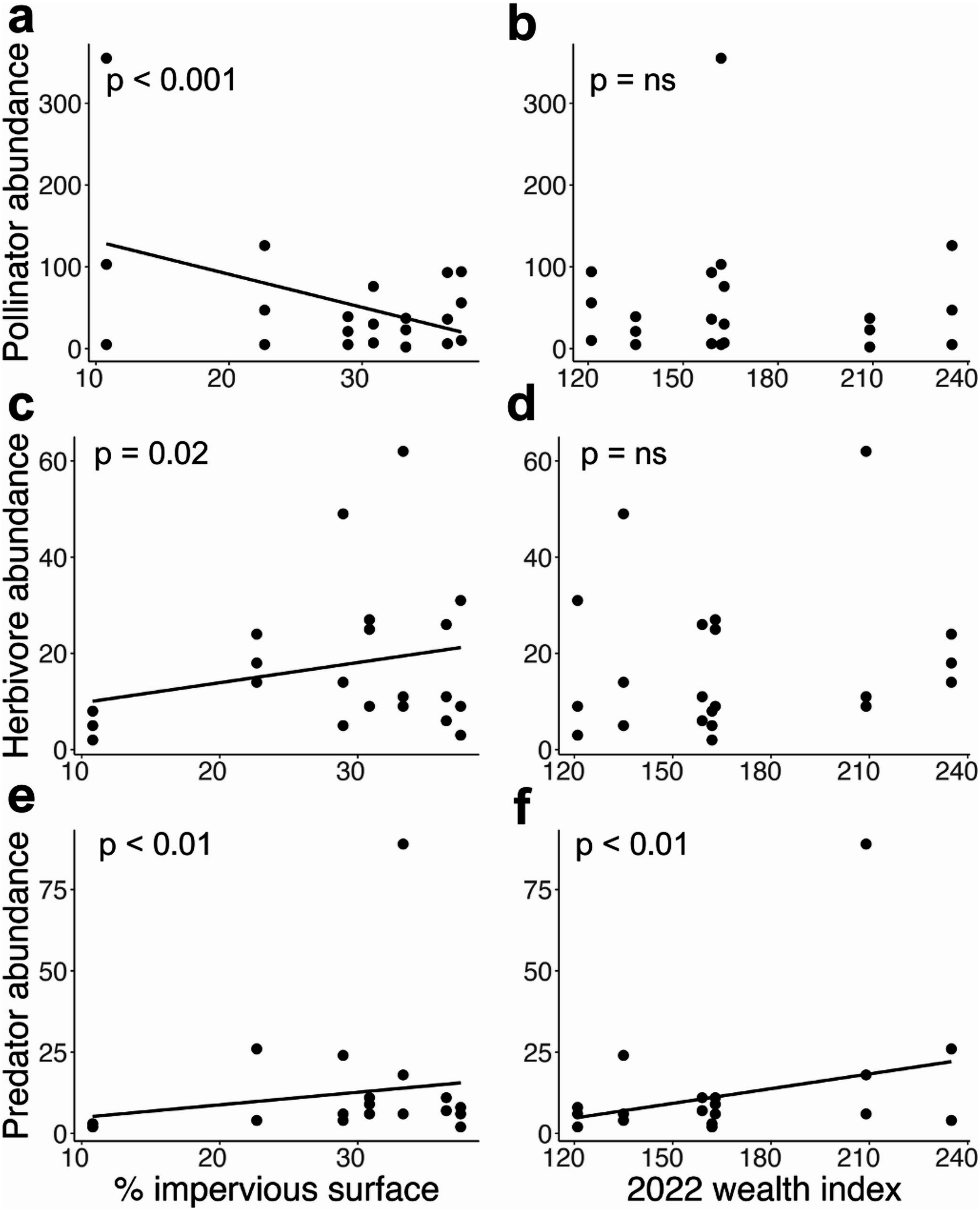

Fig. 2 Relationships between percent impervious surface and site wealth index and (**a & b**) pollinator abundance ($R^2_{GLMM\ (m)} = 0.08$; $R^2_{GLMM\ (c)} = 0.84$), (**c & d**) herbivore abundance ($R^2_{GLMM\ (m)} = 0.15$; $R^2_{GLMM\ (c)} = 0.55$), and (**e & f**) predator abundance ($R^2_{GLMM\ (m)} = 0.33$; $R^2_{GLMM\ (c)} = 0.48$) using arthropod abundance data summed across traps and across three sampling dates (June, July, August 2022). All p-values were obtained from a Negative Binomial Generalized Mixed Effect Model (GLMM), with *ns* meaning a relationship was not significant ($p > 0.05$). Marginal and conditional R^2_{GLMM} values were derived using a trigamma estimate

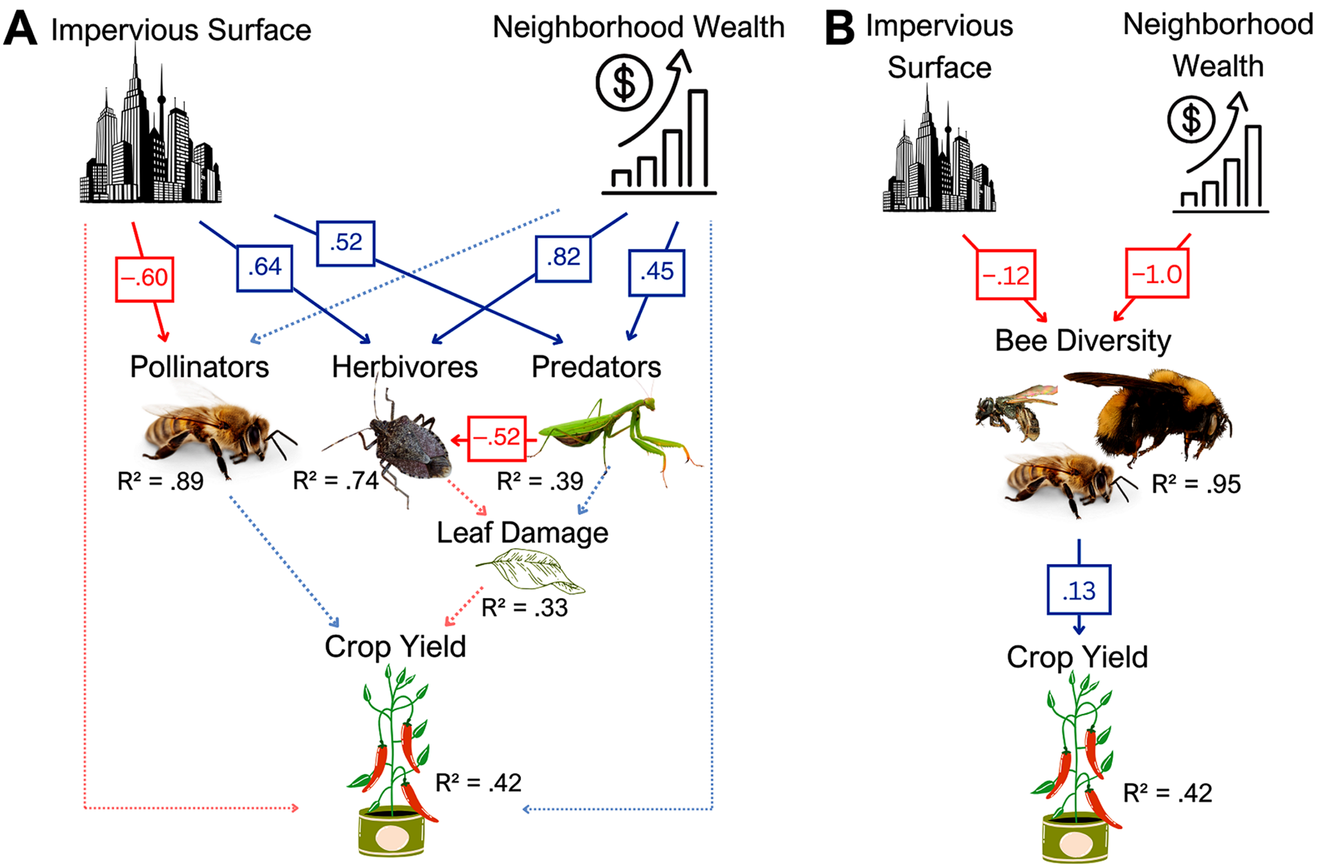

Fig. 3 Structural equation model results using data collected in August 2022. (**A**) The relationships between impervious surface and neighborhood wealth, arthropod functional group (pollinator, herbivore, and predator) abundances, and jalapenõ plant herbivory and crop yield (fruit mass). (**B**) The relationships between impervious surface and neighborhood wealth, bee Hill-Simpson diversity, and jalapenõ plant fruit mass. The numbers on the arrows are standardized coefficient estimates, and the numbers beneath the response variables show the pseudo-R^2 value for each equation. Red lines show negative relationships (also shown with negative coefficients), blue lines show positive ones (also shown with positive coefficients), and dotted lines show nonsignificant ones. Image credits: (**A**) Canva Pro License, (**B**) Canva Pro License & © DiscoverLife.org

1.04 (±0.02 SE) increase in herbivore individuals ($z=2.32$, $p=0.02$, Fig. 2c) and a 1.07 (±0.02 SE) increase in predator individuals ($z=2.66$, $p<0.01$, Fig. 2e). Herbivore abundance for the summer was not significantly related to wealth index ($z=1.92$, $p=0.055$, Fig. 2d). Predator abundance was significantly positively related to site wealth index (1.69 ± 0.27 SE, $z=2.99$, $p<0.01$, Fig. 2f).

Bee diversity

As the wealth index around sites increased, bee-Hill Simpson diversity decreased (-0.15 ± 0.06 SE, $t = -2.34$, $df=16$, $p=0.03$). Because bee Hill-Simpson diversity was rarefied, it was not highly correlated with pollinator abundance at a site ($r=0.072$), verified by a Pearson product correlation test ($t=0.403$, $df=31$, $p=0.69$). There was no significant effect of impervious surface surrounding garden sites on bee Hill-Simpson diversity for the full summer community ($t = -0.13$, $df=16$, $p=0.89$). There was no significant effect of impervious surface ($R^2=0.23$, $p=0.61$) and a significant effect of wealth index ($R^2=0.81$, $p=0.046$) on bee community composition after running a permutation test with the ordination axes from our NMDS (NMDS plot shown in Fig. S5).

Structural equation model results

Our model had a Fisher-C global goodness of fit value of 24.48 with 20 df ($p=0.22$), indicating no significant relationships between unconnected variables in our model. No significant relationships existed between our environmental covariates (wealth index and impervious surface) and our experimental pepper plant responses (leaf damage and fruit mass). Overall, the SEM results (Fig. 3a, full model output in Table S3) indicate that **(a)** pollinator abundance was significantly negatively related to impervious surface ($R^2=0.89$, $p<0.001$) but not significantly related to neighborhood wealth ($R^2=0.89$, $p=0.13$), **(b)** herbivore abundances were positively related to both impervious surface and neighborhood wealth ($R^2 = 0.74$, $p < 0.001$ and $p < 0.001$), **(c)** predator abundances were positively related to both impervious surface and neighborhood wealth ($R^2 = 0.39$, $p = 0.038$ and $p = 0.036$), and **(d)** predator and herbivore abundances were negatively correlated with each other ($R^2 = 0.74$, $p < 0.001$). No other pathways in the model were significant, including relationships between arthropod abundance, leaf damage, and pepper mass. Both neighborhood wealth and impervious surface were negatively correlated to bee Hill-Simpson diversity ($R^2 = 0.95$, $p < 0.001$ and $p = 0.03$). Bee Hill-Simpson diversity was positively correlated with experimental pepper plant mass. As Hill-Simpson diversity increased, the ln pepper mass increased at a rate of 0.126 (±0.048 SE) grams ($R^2 = 0.42$, $p = 0.016$, Figs. 3b and 4).

Discussion

We found that the abundance of pollinator, herbivorous, and predatory arthropods and bee diversity in urban community gardens was related to the amount of impervious surfaces and neighborhood wealth values in the landscape surrounding a garden, with different functional groups exhibiting different responses (Figs. 2 and 3). Additionally, our results show that bee biodiversity did not reflect the luxury effect (Fig. 3b), as bee biodiversity was negatively correlated with neighborhood wealth in this system. Finally, we found that jalapeño fruit mass was higher at sites with higher bee diversity (Fig. 4).

Herbivores, predators, and pest-control services

We predicted that predators would be negatively related to impervious surfaces because of reduced habitat and diminished matrix quality (Burkman and Gardiner 2014), which would then cause herbivorous pest abundance to increase. As impervious surfaces increased, herbivore abundances increased as predicted, but predator abundances also increased, contrary to expectations (Figs. 2 and 3a). These results match a study conducted in Los Angeles, CA, which saw a positive relationship between impervious surfaces and predator arachnid abundance (Lewthwaite et al. 2024). Another study examining aphids and their arthropod predators in urban landscapes saw that predators were negatively related to impervious surfaces but also positively related to aphid prey populations (Rocha and Fellowes 2018). These results suggest that certain predator taxa may be more sensitive to local factors, such as prey populations and local habitat complexity, than to features of the urban matrix.

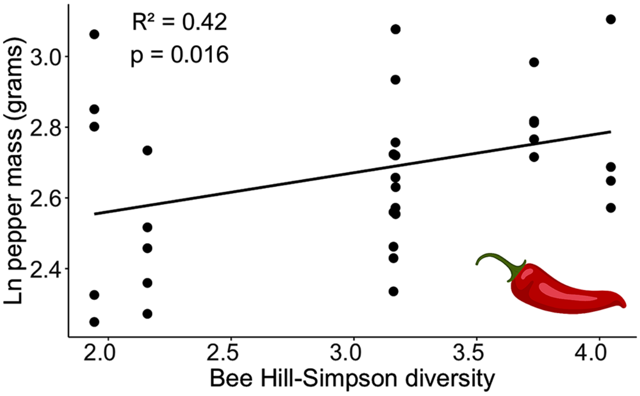

Fig. 4 The relationship between the bee hill-Simpson diversity index at a garden in August and the natural log-transformed average mass of peppers on uncovered jalapeño plants ($R^2 = 0.42$, $p = 0.016$). Image credits: Canva Pro License

Predator abundances were positively related to neighborhood wealth, as predicted. There was no relationship between herbivore abundance and neighborhood wealth across the full summer, but our structural equation model detected a positive relationship in August. This suggests that there could be important changes in insect community composition, even within guilds, throughout a season. While we did see a negative relationship between predator and pest abundances in this study, we cannot determine whether the relationship was causal (with predators reducing prey) and thus impacting pest-control services. We found no differences in leaf damage (used as a proxy for herbivory) among our sites or between our covered and uncovered jalapeño plants. We believe that there was higher than expected herbivory in our covered plants because large orthopterans (grasshoppers & katydids) were able to bite through the mesh (Asia Kaiser, personal observation). Our sampling methods were not targeted toward collecting these insects, so we are unsure how their abundances differed across our sites. This represents a limitation in our herbivore exclosure treatments, making us unable to estimate an effect size for herbivore leaf damage within a site. However, we still observed no differences in leaf damage across our sites related to herbivore abundances across the gardens (Fig. 3a). These results suggest that the abundance of herbivorous insects may not be a good predictor of herbivory pressure on plants in an urban agroecosystem. A more effective measure may be to look at the abundance of the most highly destructive herbivore taxa.

Pollinator abundance, bee diversity, and pollination services

Both pollinator abundance and bee diversity decreased as the impervious surfaces around sites increased. These results are consistent with numerous studies examining the effect of urbanization on wild bee assemblages (Ahrné et al. 2009; Matteson et al. 2012; Geslin et al. 2016). This suggests that the habitat area provided in community garden sites alone is insufficient to support diverse bee populations when there are high amounts of impervious surfaces in the surrounding matrix. While there was no relationship between pollinator abundance and neighborhood wealth, there was a significant negative relationship between bee diversity and neighborhood wealth. This was the opposite of what is predicted by the luxury effect (Hope et al. 2003; Leong et al. 2018). This *reverse* luxury effect for bee diversity was especially surprising for our study system in Boulder, CO, as a past meta-analysis found that arid landscapes exhibit even stronger luxury effects than non-arid landscapes (Chamberlain et al. 2020).

A potential explanation for this discrepancy is that very few studies have looked at luxury effects on arthropods, with 95% of studies in the meta-analysis focusing on plants or birds (Chamberlain et al. 2020). The nesting behaviors of bees may also be a potential mechanism for why neighborhood wealth is negatively correlated with bee diversity. Most (> 70%) bee species are ground nesting (Antoine and Forrest 2020; Harmon-Threatt 2020) and prefer bare ground for nesting spots (Gardein et al. 2022). Abiotic factors like mulch have been shown to reduce pollinator diversity and abundance (Cohen et al. 2020). If higher-income neighborhoods have more manicured lawns with more irrigation, turf grass, mulch, and pesticide use (Fraser et al. 2013; Fuentes 2021), this may deter most bee species from nesting in these areas. Around 94% of the bees we sampled were ground nesters, which means these mechanisms are highly relevant to this bee community (Table S1). One study looked at the cultural drivers of yard landscaping decisions by college-educated homeowners surrounding community garden sites with consistently low pollinator scores and found that many lawns had low vegetative complexity due to a perceived socio-cultural pressure to maintain a “perfect lawn” (Burr et al. 2018). Additionally, wealthy areas tend to have more moist environments from irrigation and tree cover, but bee biodiversity is richest in arid and semi-arid environments (Orr et al. 2020).

Understanding the impact of environmental covariates on bee diversity may be especially important for urban food production, as bee Hill-Simpson diversity was the only variable with a positive relationship to jalapeño pepper mass in our study. This is supported by a previous urban agriculture study in California, which found that jalapeño pepper seed count increased as pollinator richness increased (Cohen et al. 2020). In our study, pollinator abundance alone was not a significant predictor of fruit size. This shows that increasing pollinator population sizes alone may not be enough to ensure robust pollination services, and maximizing pollinator species richness must also be prioritized to increase pollinator community effectiveness (Willcox et al. 2017; Woodcock et al. 2019).

Study limitations

A limitation of this study is that the arthropod exclosures on our sentinel plants did not work as intended, so we could not achieve a causal estimate of the pollination and pest-control services provided at each site. While we observed differences in pepper mass across our sites, no differences existed between our covered and uncovered plants. We believe this may be because the mesh coverings changed environmental variables affecting the plant growth unrelated to arthropods. Although the mesh used was very lightweight, it could have provided additional shading to plants and slowed down evapotranspiration. Additionally, while jalapeño plants have been shown to benefit from insect pollination (Cohen et al. 2020), they do not require it, and perhaps wind within covered plants may have been sufficient to compensate for the lack of pollinator visitation. As mentioned above, certain herbivores were able to bite through the mesh exclosures as well, causing us to see no differences in leaf damage between covered and uncovered plants at a site.

Another potential limitation of this study is using passive sampling methods (pitfall and vane traps) instead of active sampling methods such as aerial nettings. While traps catch many more insect specimens, active sampling methods are better at capturing rare species (Montgomery et al. 2021). Passive sampling may have caused highly effective pollinators and natural enemies or deleterious pests to be underrepresented in our sample, despite them having a large impact on our jalapeño plants. For example, we observed in the field (and numerous gardeners confirmed) that invasive Japanese beetles (*Popillia japonica*) were highly abundant and caused significant crop damage. However, we collected relatively few individuals ($n = 11$). This may be because residents in the surrounding neighborhood were using baited Japanese beetle traps (Asia Kaiser, personal observation), making them less likely to be caught in our traps. As a result, passive sampling methods may not always provide a true representation of the arthropod community composition and should be complemented with active sampling methods in future research.

A third limitation of our data is that our relatively small number of sites may not have captured the full range of socioeconomic values in our neighborhood wealth values. Our dataset’s wealth values range from 121 to 235 (with 100 representing the national average), meaning we are missing data on the lowest-income neighborhoods. The luxury effect in certain arthropod groups may also be non-monotonic, and biodiversity decreases above and below certain socioeconomic levels. A past study looking at bird species richness in the developing world similarly found a negative relationship between wealth and diversity in highly urban neighborhoods, but the relationship flipped in less urban landscapes

(Chamberlain et al. 2019). Although floral diversity is sometimes higher in high-income neighborhoods (Leong et al. 2018), perhaps floral community composition or other landscaping practices, like irrigation, fertilizer, and mulch use, differ in a way that reduces the abundance and diversity of certain arthropod groups.

Conclusions

Community gardens support arthropod populations in urban areas, and subsequently, those arthropods provide essential pollination and pest control services to the humans utilizing these spaces. As the world becomes increasingly urban (United Nations, 2018), it will be important to understand how to maximize these benefits for humans and arthropods in urban environments. Our results show that larger landscape environmental features like impervious surfaces negatively correlate with pollinator abundance and bee diversity but positively correlate with herbivore and predator arthropod abundances within community garden sites. However, wealth indicators alone may not be sufficient to predict the urban matrix quality surrounding community gardens. High-resolution Urban Land Use Classification tools may better predict matrix quality and, thus, the arthropod populations and the ecosystem services they provide at community gardens (Cadenasso et al. 2007). Previous studies have found that management actions like increasing plant species diversity, including woody vegetation at sites, and increasing garden size can increase pollinator abundance and diversity at community garden sites (Gaston et al. 2005; Majewska and Altizer 2019). Additionally, increasing habitat complexity in urban gardens by including more trees, shrubs, and plant species has been shown to increase the diversity and abundance of insect predators and the pest control services they provide (Philpott et al. 2020). The results of our study show that locating gardens in areas with less impervious surface or improving habitat in the surrounding matrix can increase bee abundance and diversity and, subsequently, pollination services. Future research should focus on understanding the mechanisms driving the environmental differences between neighborhoods with different socioeconomic conditions, as the luxury effect may be an overly simplistic explanation for biodiversity patterns in cities. This research will inform management actions at garden sites and in surrounding neighborhoods to ensure equitable quality of pollination and pest control services.

Statements and Declarations.

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Declarations

Competing interests The authors declare no competing interests.

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