Urban Ecosystems (2026) 29:100
https://doi.org/10.1007/s11252-026-01953-1

RESEARCH

Suburban land management impacts bee community composition and cucumber (*Cucumis sativus*) pollination

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Received: 16 October 2025 / Accepted: 7 March 2026 / Published online: 30 March 2026
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Abstract
Urban green spaces can provide quality habitat for insect pollinators. However, few studies have investigated the quality of pollination services provided by urban insect pollinator communities or compared pollination services between different types of urban green spaces. We evaluated both the taxonomic and functional trait diversity of flowering plant and bee communities in two common types of suburban green space: residential backyards and electric powerline corridors. We also quantified seed set from sentinel cucumber plants as a measure of pollination services. We analyzed the response of the flowering plant community to site type and impervious surface cover, as well as the response of the bee community to flowering plant taxonomic and functional diversity and impervious surface cover. Seed set was analyzed in response to bee abundance, species diversity, and the community-weighted mean of bee body size. Site type affected floral resource diversity. Bee abundance was negatively affected by both impervious surface cover and floral resource functional evenness, and pollination services were negatively affected by bee total abundance. Our findings suggest that certain suburban management practices and site characteristics can lead to bee communities with highly abundant but mostly ineffective pollinators. Ultimately, these results also reinforce the need for more urban ecology studies that directly measure ecosystem functions and attempt to trace the mechanistic relationships between those functions and the organisms facilitating them.

Keywords Urban ecology · Ecosystem function · Ecosystem service · Pollinator · Land management

Introduction

Urban green spaces can provide important habitat for insect pollinators (Hall et al. 2017, Langellotto et al. 2018; Dylewski et al. 2020), and some studies have found that urban areas can harbor higher bee species richness and diversity than rural (Wilson & Jamieson 2019), natural (Banaszak-Cibicka et al. 2018), or semi-natural areas (Martins et al. 2017). However, urban green spaces vary in their habitat types and qualities (Rebele 1994), as well as their urbanization intensity, which is often quantified by measures like impervious surface cover (Buchholz et al. 2020). Characteristics of urban areas including temperature, population density, impervious surface cover, management intensity, and cultural norms can all act as filters on the communities of organisms that are able to live in those areas (Aronson et al. 2016; Fenoglio et al. 2021). Some scientists have argued that any research on urban green spaces should compare multiple types against one another (Ayers and Rehan 2021) or against a natural landscape control (Filazzola et al. 2019) rather than examining a single green space type in isolation. Multi-type studies of urban green spaces provide an opportunity to compare the effects of management on vegetation, insect pollinator communities, and pollination services.

Because pollinator conservation is under threat worldwide, more researchers have focused on pollinator diversity and activity in urban areas, especially bees (Burr et al. 2016; Daniels et al. 2020). Although many arthropods, birds, and other animals are important pollinators, the majority of

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pollination research, including the present study, is focused on bees due to their abundance, relative ease of identification, and large contribution to pollination services. One urban green space type that has received attention in pollinator research is the residential yard or garden, but results differ on whether gardens are more (Baldock, et al. 2019) or less (Dylewski et al. 2019) beneficial for pollinators than other urban green spaces. This lack of consensus may reflect differences in yard and garden management that affect the floral resources available to pollinators (Majewska & Altizer 2020). For example, Lerman et al. (2018) found that varying the frequency of lawn mowing altered the number of flowers in lawns, which altered bee abundance and species richness. Homeowner choice of plantings can also affect the quality of floral resources, as the majority of floral resources in most residential gardens are ornamental non-native plants (Lowenstein & Minor 2016, Baldock, et al. 2019). As floral diversity, abundance, and composition have been closely linked to attributes of the bee community, the choices people make in their yards can have important implications for pollinator conservation. Most pollinator research has been conducted in front yards, leaving backyards relatively under investigated -- even though homeowners manage their front and back yards differently (Larsen and Harlan 2006; Locke et al. 2018). Therefore, the effects of homeowner backyard management and plant choice on pollinators requires further investigation.

Powerline corridors, which frequently crisscross urban and suburban areas, have also received attention from the pollinator research community (Dániel-Ferreira et al. 2020; Russo et al. 2021, Steinert et al. 2020). Management of many powerline corridors in North America is similar to that of residential yards, including mowing and herbicide use (Wojcik and Buchmann 2012), although there is very little research comparing the two types of urban green spaces. Multiple studies have found that the semi-natural grassland habitat and floral resources provided by powerline corridors can support more abundant and diverse communities of bees compared to the surrounding landscape (Hill & Bartomeus 2016, Hosler et al. 2025; Wagner et al. 2019). However, whether these bee communities actually provide sustainable levels of pollination services is yet to be determined (Wojcik and Buchmann 2012).

Pollination services can be evaluated by quantifying the fruit and/or seed set of sentinel plants, which in application are target plants observed as "indicators" for a variety of environmental functions and disturbances (Lower and Kendall 1990). Steffan-Dewenter and Tscharntke (1999) found that bee abundance best predicted fruit and seed set in focal plants, while other studies indicate that pollinator species richness is a better predictor of fruit and seed set than pollinator abundance (Lowenstein et al. 2015; Cohen et al. 2020). While some previous studies have used pollinator visitation as a proxy for pollination, visits can vary widely in quality (Greenop et al. 2023), and duration of a pollinator's visit does not correlate with pollen deposition (Thomson & Goodell 2001). Variation in pollination outcome may occur due to the taxonomic identity of the flower visitor and/or the type of foraging visit (i.e. collecting nectar vs. collecting pollen; Thomson & Goodell 2001). Other studies have demonstrated that the species richness or diversity of flowering plants near a sentinel plant will increase pollinator visitation and seed set (Hennig & Ghazoul 2011, Cohen et al. 2020; McDougall et al. 2022), so studies of pollination services should be most effective if they evaluate the pollinator community, study area characteristics, and a direct measure of pollination such as seed set.

Historically, taxonomic approaches (e.g., species richness, Shannon diversity, etc.) have been employed to evaluate diversity of important taxa like pollinators. But recently, researchers have suggested that functional trait-based approaches may be more useful when investigating the ecosystem services (Pinho et al. 2021) provided by green infrastructure and rights-of-way (Filazzola et al. 2019). Trait-based approaches can provide a better understanding of complex ecological systems and relationships than traditional taxonomy-based approaches (Violle et al. 2007; Gagic et al. 2015), but there is still disagreement over which method is most effective (Garibaldi et al. 2015; Hatfield et al. 2018). Two common approaches are well-suited for trait-based studies of groups like pollinators: metrics describing the mean values of *individual* traits and metrics describing the variation in *multiple* traits. Functional identity metrics, or community-weighted means (Gagic et al. 2015), emphasize the average community values for individual traits such as body size. Alternatively, complementarity metrics such as functional richness (range of trait values; Mason et al. 2005), functional evenness (evenness of trait value distribution; e.g. niche complementarity; Mason et al. 2005), functional divergence (how the distribution of abundances in trait space maximizes total community variation; Mason et al. 2005), and functional dispersion (mean distance to abundance-weighted community centroid; Laliberte and Legendre 2010) evaluate the distribution and abundance of multiple traits across the community.

Functional diversity metrics can include response traits, effect traits, or both (Violle et al. 2007). Response traits are organismal characteristics whose values respond to environmental variation. For example, plant species' native (or non-native) status can be considered a response trait as it can be filtered by environmental conditions; native plant species may be chosen in the case of an actively-managed restoration, while non-native plant species with certain life strategies may be selected for in the case of a disturbed fallow site

(Laughlin 2014, Swan et al. 2021). Conversely, effect traits are organismal characteristics that directly affect ecosystem functions. Bee hairiness is an example of an effect trait, as the amount of hair on a bee directly affects how much pollen will be transported by that bee between flowers (Stavert et al. 2016), therefore affecting the ecosystem function of pollination. Some species traits, like plant height, can be both response and effect traits: plant height responds to environmental filters such as nutrient availability and precipitation (both via intraspecific variation and environmental filtering of species), but plant height also affects multiple ecosystem services including primary production, carbon sequestration, and even pollination, by affecting visitation by pollinators (Cornelissen et al. 2003). Floral size could also be considered both a response and effect trait: species with larger blooms may be selected for residential gardens for aesthetic reasons, while the size of blooms can also affect which pollinators are attracted to a plant (Blarer et al. 2000). Recent studies have shown that some pollinator functional traits, especially morphological traits like hairiness (affects the amount of pollen stuck to the body), diet specialization (determines which plant species are likely to be visited), body-size (a commonly used proxy-trait), and inter-tegular distance (allometrically related to proboscis length; Cariveau et al. 2016), are strong predictors of pollen deposition by flower visitors (Földesi et al. 2021, Stavert et al. 2016). It is possible that functional trait-based approaches may be stronger predictors of pollination than measures of taxonomic diversity or visit frequency or duration, indicating that a trait-based approach may be more appropriate than a taxonomic approach when trying to understand ecosystem service provisioning.

Some studies have attempted to connect the functional traits of pollinator communities to pollination services through sentinel plant assays. Studies using the "functional group" approach lump pollinators into categories such as social bees, solitary bees, and hover flies (Albrecht et al. 2012) or *post hoc* guilds determined by gross body size (i.e. "very small," "small," "medium," etc.) and behavior (Hoehn et al. 2008), rather than using specific trait values and indices calculated from those values. These studies found that a diversity of functional groups increased fruit and/or seed set (Hoehn et al. 2008; Albrecht et al. 2012).

More recent studies have also shown that pollinator community functional richness (Fründ et al. 2013), functional divergence (Woodcock et al. 2019), Cunningham-Minnick et al. 2020), or functional dispersion (Martins et al. 2015) can have positive relationships with sentinel plant fruit and/or seed set. However, many studies investigate only one metric of functional diversity, and the effect of functional diversity on pollination function has rarely been examined in an urban context. Schwarz et al. (2017) noted few studies directly connecting biodiversity to ecosystem functioning in urban areas, and of the studies that did use empirical methods, a minority considered functional traits. Therefore, more empirical studies to directly investigate both pollinator functional diversity and pollination function in urban areas are needed.

Here we investigate how the bee and flowering plant communities vary between suburban green spaces managed in different ways, and how that variation affects pollination services in those areas. We surveyed bees in suburban residential neighborhoods and electric transmission powerline corridors, as these two green spaces both receive direct and often regular management and together comprise a large portion of the Greater Chicago Area. We quantified the taxonomic and functional trait structure of these communities and measured in situ pollination services to better understand how suburban green space management shapes communities and ecosystem function. Our study addresses three research questions: (1) Does site type explain variation in different measures of the flowering plant community? (2) What characteristics of the flowering plant community affect the bee community? and (3) What characteristics of the bee community best explain variation in seed set of sentinel plants? We expected (1) site types with lower management intensity (i.e. prairie and oldfield powerline corridors) will harbor more taxonomically and functionally diverse flowering plant communities than sites with more intense management (i.e. mown powerline corridors and residential yards); (2) more diverse flowering plant communities will lead to more diverse bee communities; and (3) more diverse bee communities will lead to greater seed set, but with functional diversity indices showing stronger patterns than taxonomic indices. Our study serves as a unique and important contribution to both the urban pollination and biodiversity-ecosystem services bodies of literature.

Materials and methods

Study Area

This study was conducted in powerline corridors and residential neighborhoods across the Chicago, IL metropolitan area (Cook, DuPage, and McHenry counties) during the summer of 2021. The powerline corridors are managed by the Commonwealth Edison company ("ComEd"), the primary provider of electrical power in the area. ComEd's high voltage powerline corridors cover over 22,945 hectares in Illinois and are managed in multiple ways.

Four study clusters were selected near the communities of Crystal Lake, Glendale Heights, Morton Grove, and Naperville, IL USA (Fig. 1A). We chose these clusters to represent known geographic and demographic differences across the

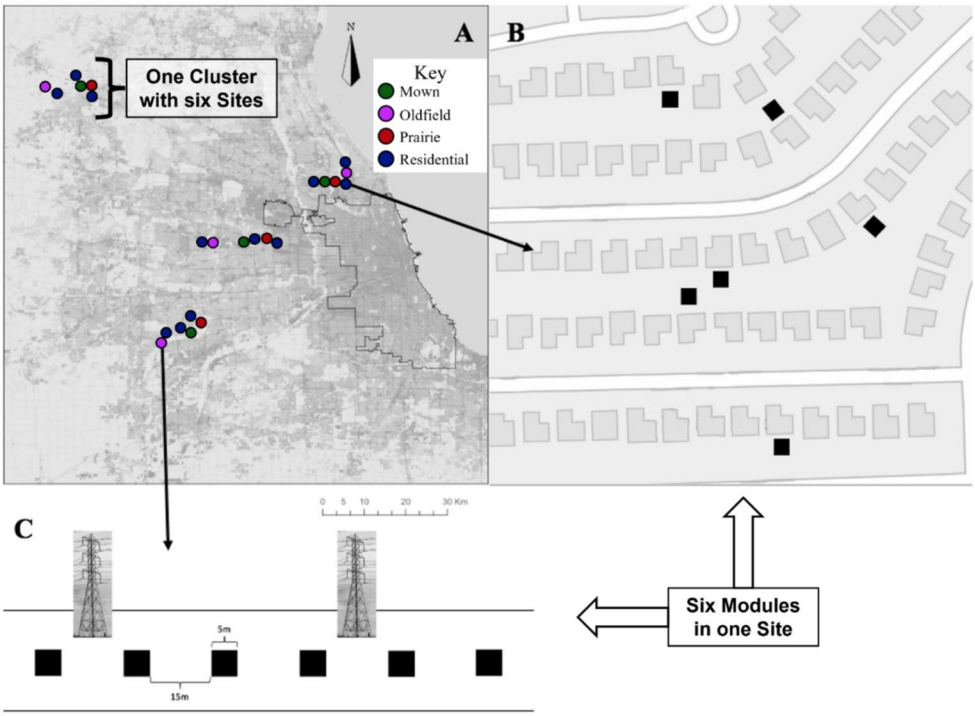

Fig. 1 (**A**) Map of the Chicago Metropolitan Area, IL USA; orange indicates impermeable surface cover and the black line delineates the city of Chicago. Circles indicate study sites arranged in 4 clusters with 6 sites each. (**B**) Map of a single residential study site (neighborhood) with backyard modules marked as black squares. (**C**) Diagram (not to scale) of a single powerline corridor site with modules marked as black squares. Map A was created by M. Garfinkel, Map B is a screenshot from Google Maps, and the photos in C were captured by S. C. Hosler

Chicago area. The clustering approach facilitated comparison of corridor types, as prairie corridors are relatively rare. In each study cluster, we chose three corridor sites based on management type: one prairie corridor (native vegetation), one oldfield corridor (volunteer vegetation), and one mown corridor (turfgrass). We also selected a residential neighborhood near each of the three corridor sites. Collectively, the three corridor sites and the three nearby neighborhoods comprise a single study cluster.

"Prairie" corridors were either remnant prairie patches adjacent to or within the boundaries of a county forest preserve, or areas where ComEd was working with stakeholders to restore native prairie vegetation. "Oldfield" corridors were managed on a 5-year rotation, with large woody vegetation removed in years 1 and 2 and the remaining vegetation left relatively untouched in years 3–5. "Mown" corridors were characterized by relatively short turfgrass mowed several times during the growing season. All powerline corridor sites were geographically close in proximity to the other corridor and neighborhood sites within a cluster in order to reduce potential confounding effects of nearby habitats, weather, and soil type. We also ensured that each powerline site was bordered by single-family homes on at least one edge in order to control for effects of adjacent land cover. Corridor sections within each cluster were located at least 1 km from one another to reduce potential overlap of bee communities. This distance was chosen based on estimates of urban bee flight distances (Langellotto et al. 2018).

Within each neighborhood, focal yards were selected through a process of door-to-door visitation to receive homeowner permission. Here we use the term "yard" to refer to the outside area surrounding a home. In our study region, this outside area is often dominated by mowed turfgrass but also can include flower or vegetable gardens, woody vegetation, mulched areas, or paved areas. Residential yards are managed according to the preferences of each homeowner, which can contribute to high habitat heterogeneity in any one neighborhood (Hosler et al. 2025). However, at a larger scale, and in comparison to natural habitats, residential yards are quite homogeneous (Ignatieva and Hedblom 2018), as many homeowners in Chicago use the same popular ornamental plant species in their landscaping (Hosler et al. 2025). The number of participating yards in each neighborhood varied from 4 to 7 due to homeowner withdrawals and voluntary additions to the study during the first sampling session, with a total of 68 yards included in the study.

As suburban density varies across the Greater Chicago Area, we also measured impervious surface cover in a 1 km buffer surrounding the centroid of each of our study sites (Buchholz et al. 2020). We downloaded recent impervious surface land cover layers from the National Land Cover Database (NLCD, Dewitz 2023) and used QGIS software version 3.22.16 (QGIS Association) to create the 1 km buffers and extract the data for each site (Appendix S1). We ran a linear model on the log-transformed impervious surface data (R software version 4.2.2; R Development Core Team

2025) to determine whether impervious surface cover varied by geographic cluster of our research sites and therefore whether we needed to include impervious surface or geographic cluster as a factor in our subsequent models. Model results for impervious surface cover can be found in Appendix S1.

Flowering plant and bee sampling design

The area of a residential yard (both front and back) differs widely among homes in the greater Chicago area owing to differences in lot size and house footprint, and the presence/absence of other structures on the property (e.g., patios and porches, storage sheds, swimming pools, etc.). Additionally, the width of powerline corridors in the study area varies due to zoning and geographic features including streams, forest preserves, etc. We therefore used 25 m^2 modules as a standardized means of sampling flowering plants, bees, and pollination function in all of our study sites. We established one 25 m^2 sampling module within each backyard, with the four to seven participating yards per neighborhood being treated as one neighborhood "site" equivalent to one corridor site (Fig. 1B). Placement of residential sampling modules was established to include some non-lawn garden area (if present) and some lawn/turfgrass area (if present) and to minimize impermeable surface area (e.g., sidewalks, patios, driveways). Several backyards were arranged in such a way that we could not establish our sampling modules in squares. In these situations, we split the 25 m^2 area into two rectangular sections and established them according to the same guidelines.

In corridor sites, we established six 5 m × 5 m modules separated by a minimum of 15 m and arranged in a relatively linear sequence (Fig. 1C). Arrangement and spacing of modules within corridors varied rarely, with variation due to spatial limitations within the corridor (e.g., placement of powerline towers, proximity of adjacent waterways, presence of atypical vegetation sections due to adjacent homeowner mowing). We sampled the flowering plant and bee communities at each module twice, once in early summer (within three weeks in late June/early July) and once in late summer (within three weeks in late August/early September).

Flowering plant species and traits

We surveyed each sampling module for flowering plants concurrently with bee sampling (see below), with flowering plant surveys being conducted on the first day of each sampling visit at a site. A single observer performed all flowering plant surveys by counting the visible open floral units on all forbs and shrubs (but not grasses or trees) within the module and identifying them to the lowest possible taxonomic level, using the photo-recognition capabilities of (iNaturalist 2021) when needed. A "floral unit" was considered to be a single flower (e.g. roses) or an inflorescence (e.g., an umbel, like wild carrot; a spike, like salvia; or a composite flower, like daisy), depending on which unit would most logically be called a "flower" by a layperson (as data from these surveys were also incorporated into a different study regarding residents' preference for different types of powerline corridor management; Garfinkel et al. 2023). In the case of our identification procedure, "lowest possible taxonomic level" included both full Latin binomial species names when possible, but also genus names when an exact species could not be determined (as was the case for many cultivated ornamental plants).

We characterized each flowering plant species by a set of morphological, phenological, and life history traits. These included average plant height, flower primary color (in the visible spectrum), flower size (measured as diameter or length of a floral unit), plant native status, and plant life cycle (annual, biennial, or perennial). Trait values were collected from the literature (Dickinson and Royer 2014; Wilhelm and Rericha 2017). See Appendix S2 for a full list of species and trait values. Because traits may be related and therefore provide redundant information, we used the Variance Inflation Factor (VIF; Akinwande et al. 2015) to test for multicollinearity among our flowering plant species trait values. All of our flowering plant species traits returned VIF values < 3 during the initial VIF test, so we included all of the traits in subsequent analyses.

Bee community and traits

Sampling for both flowering plants and bees was conducted on the same day for all sites within a single geographic cluster, between the hours of 8 am and 7 pm when the air temperature was above 20° C, wind speeds were less than 25 km/h, and conditions were partly cloudy to sunny.

We sampled the bee community at each module with timed observations and pan traps. These complementary methods were used in combination to achieve a more complete survey of the bee community, as visual observations are biased toward large-bodied species, whereas pan traps are more likely to capture only small-bodied species (Wagner et al. 2014). Numerous recent studies have established a precedent for combining bee samples collected from multiple sampling techniques (Gerner and Sargent 2022; Langellotto et al. 2018; Lerman et al. 2018; Turo et al. 2021). As the goal of our study was to identify relationships between the bee community, flowering plants, and pollination services, we felt it would be most appropriate to combine the data from these two sampling techniques rather than to

calculate separate indices of functional diversity for each sampling method. Prior to combining the data from the two sampling methods, we standardized the data by dividing the abundance of each species in each module by the total number of bees captured using that sampling method (following Hosler et al. 2025). Data standardization was conducted because the two sampling methods used are each better for capturing different species and may yield very different abundances of individuals.

During timed observations, a single observer stood in each module and observed all visible bees active within it (a modified Pollard walk; Pollard 1977). As some residential modules were split in two, the primary observer dedicated 5 min to each half of a module, for a total of 10 min of observation per module (40 min to 1 h total observation time per site depending on the number of modules). The observer had experience identifying bees in flight in the Chicago area and used a region-appropriate set of keys for field identification. Individual bees were identified to the lowest possible taxonomic level. As with flowering plants, "lowest possible taxonomic level" included both full Latin binomial species names when possible, but also genus names when an exact species could not be determined (as was the case for some individuals that moved too quickly for precise identification, e.g. *Bombus spp.*). When necessary and possible, individuals were captured with an aerial net by a second observer, placed in a lidded petri dish, and photographed for identification using iNaturalist before being released.

To supplement the timed observations, we also deployed arrays of pan traps in a method similar to that of Wagner et al. (2014). We used small plastic dessert bowls (10 cm diameter) in three colors (blue, white, and yellow) and deployed six per module (two of each color). Pan traps were partially filled with a dilute unscented dish soap solution and placed flat on the ground to maintain trap height consistency across sites with varying groundcover (e.g., mown turfgrass versus mulch versus bare ground versus tall forbs). In residential modules that were split in two, three bowls (one of each color) were placed in each rectangular section. Otherwise, in square residential modules and all corridor modules, the six bowls were placed along a randomized diagonal transect between two module corners with no two bowls of the same color adjacent to one another. We deployed the pan traps immediately following the timed observations in each module, then collected them approximately 24 h later. During collection, contents of all six bowls were strained and combined into a labeled container filled with 70% ethanol. In the lab, samples were hand-sorted by a single observer, and all bees were washed, dried, and pinned. Samples were identified to lowest possible taxonomic level using iNaturalist, Discover Life, and Arduser (2020), with *Lasioglossum* individuals being grouped into morphospecies based on obvious morphological differences (e.g. coloration) due to the difficulty of accurately identifying members of this genus to species.

We selected a set of bee morphological effect traits (organism traits that affect ecological function) related to the mechanics of pollination. These included body size (measured as body length), intertegular distance, tongue length, pollen-carrying structures (absent, corbicula, scopa on tibia, scopa on femur, scopa on both tibia and femur, or scopa on abdomen), and hairiness (estimated on a scale from 0-no hairs to 5-completely covered in long hairs). Trait values were collected from the literature when available (BugGuide, Cariveau 2016, Discover Life, Hoebeke & Wheeler 1999, McCravy et al. 2020, Mitchell 1960, Mitchell 1962, Roquer-Beni 2020). When trait values were unavailable from the literature, measurements were taken from pinned specimens and averaged (up to 10 randomly-selected individuals or as many as were available in our collection) or assigned as a genus average from literature values (see Appendix S3). When values for tongue length were not available in the literature, we used the ITconverter() function from the BeeIT package in R (Cariveau et al. 2016) to calculate tongue length based on taxonomic family and intertegular distance. See Appendix S3 for a full list of species and trait values.

As with flowering plant species traits, we used the Variance Inflation Factor (VIF) to test for multicollinearity among bee trait values. We conducted stepwise VIF, removing variables one at a time until all remaining variables had a VIF < 3. At the conclusion of this procedure, only bee body size remained as a trait to be used in subsequent analyses.

Pollination function

To quantify pollination, we used cucumber plants (*Cucumis sativus*, Saladmore Bush Hybrid, from Park Seed, Greenwood, South Carolina, USA) in a sentinel plant assay. Cucumbers are monoecious plants with separate male and female flowers on the same plant. This means they require pollinator activity to produce seed, which makes them a common sentinel plant in ecological studies (Bennett and Lovell 2019; Cunningham-Minnick et al. 2020; Lowenstein et al. 2015). Plants were grown in self-watering pots to control for variation in soil quality between our geographically dispersed sites; the pot design was inspired by that of Cunningham-Minnick et al. (2020).

Plants were grown at the greenhouse on the campus of University of Illinois Chicago (UIC). Three cucumber seeds were planted in each pot in early May. Pots were weeded weekly, and plants were monitored for female flower buds, which emerge approximately 10 days after the male flowers. Once a majority of the pots had at least one female

flower bud, we deployed the sentinel plants to the study sites. Female flowers that opened before deployment were removed, and all female buds were marked with red tape and counted. As starting female bud counts among all pots were low, we ensured that at least one female bud was present in each module at deployment. Male flowers were not counted, as they were numerous on every plant and we did not expect our plants to be pollen-limited. Male flowers can outnumber female flowers by a ratio of 7:1 (or more) on cucumber plants (Ekeke & Agbawa 2018). Recent research in Chicago demonstrated that cucurbits do not appear to be pollen-limited in our study region (Zink et al. 2024).

Pots were deployed in mid-July. All pots within a cluster were deployed on the same day, and pots for each of the four clusters were deployed during four consecutive days. Two pots were placed in each sampling module for a total of 12 pots per site; pots were assigned random placement within corridor modules and placed with consideration to homeowner convenience in residential modules. Sentinel plants remained in the field for seven days, at which time they were collected and returned to the UIC greenhouse.

Upon collection, we marked any additional female flowers that had opened during the assay with red tape. Any unopened female buds were removed, and fruits were allowed to mature while the plants were stored on the greenhouse lawn. We harvested the fruits of the marked female flowers when they appeared ripe, 14–25 days after retrieving the pots from the sites. Cucumbers were sliced in half and their seeds were removed and frozen in individual containers until counted. For counting, each container was thawed and the contents placed in a petri dish. A single observer counted all fertilized seeds from each fruit. Our measures of *seed set* were calculated as (1) the total number of seeds produced per site, and (2) the mean number of seeds produced per cucumber per site.

Diversity measures

We pooled the flowering plant and bee community data from both sampling visits and all modules in a site (corridor or neighborhood) to calculate overall abundance of floral units and bees at each site over the entire summer. We also calculated honey bee abundance per site in addition to total bee abundance because honey bees are present on the landscape as both managed pollinators and in feral colonies. Additionally, honey bee pollination behavior can differ from that of other bee species (Kunin 1993; Westerkamp 1991).

Due to unequal sampling effort in the neighborhood sites (where some neighborhoods included as few as 4 yards and some included as many as 7 yards), we plotted rarefaction curves using the rarecurve() function from the R package vegan (Oksanen et al. 2020) to assess our flowering plant and bee community counts. Results indicated that we should use rarefied richness for both flowering plants and bees, as species accumulation curves did not consistently asymptote. We then calculated rarefied richness using the ChaoRichness() function and rarefied Shannon diversity using the ChaoShannon() function, both in the R package iNEXT (Hsieh et al. 2016). All statistical analyses were performed using R version 4.2.2 (R Development Core Team 2025).

We used the dbFD() function from the FD package in R (Laliberté & Legendre 2010) to calculate our functional diversity metrics. The FD package is capable of handling both qualitative and quantitative trait values, and addresses variation in trait value scales by computing PCoA axes and using the Gower dissimilarity coefficient in the dbFD() function (Laliberté & Legendre 2010). For the flowering plant community, we used dbFD() to calculate two functional diversity metrics: (1) functional evenness, a measure of the regularity of the distribution of abundances of individuals with different trait values within the trait space (FEve), and (2) functional dispersion, the mean distance of species in multivariate niche space from a centroid weighted by species abundances (FDis; Laliberte and Legendre 2010). We selected functional dispersion because it is considered to be an improvement on the functional divergence measure, the latter of which was identified by Mason et al. (2013) as being most often linked to ecosystem functions. However, while Mammola et al. (2021) agreed that measures of functional dissimilarity like functional dispersion are important, these authors assert that no single measure of dissimilarity can adequately represent the "richness, divergence, and regularity" components of functional diversity, so we chose to include functional evenness to complement functional dispersion and represent the "regularity" component as well. We did not calculate functional diversity metrics for bees since we analyzed only one trait (body size).

We also used the dfFD() function to calculate the community-weighted means (CWMs) of the functional traits for both flowering plants and bees. Community-weighted means are calculated as means for each trait in the community weighted by the abundances or frequencies of the values present (Gagic et al. 2015). The FD package is capable of receiving trait data matrices containing a combination of continuous and categorical data, so there was no need to transform categorical traits prior to calculation of trait-based indices. However, for the "native status" functional trait of the flowering plant species, for which there were only two factor levels (i.e. native and non-native), we transformed those trait values into binary categories with values of "0" and "1" and used the binomial distribution when modeling the Community-Weighted Mean (CWM) index for this trait.

Table 1 Summary statistics for flowering plants, bees, and cucumbers. Minimum and maximum values refer to the totals for individual study sites. Bowls and Observations refer to the two bee sampling methods

Ecological Communities		
Taxon	*Morphospecies Richness Total (min and max)*	*Abundance Total (min and max)*
Flowering Plants	156 (6–42)	36,205 (260-4,125)
	Overall: 55 (9–28)	Overall: 3,503 (19–587)
Bees	Bowls: 46 (6–21)	Bowls: 3,118 (14–587)
	Observations: 16 (0–9)	Observations: 385 (0–49)
Cucumber Sentinel Plants		
Total Fruits	*Total Seeds*	*Seeds per Fruit*
300	5,383	Mean: 17.9 Max: 257 Min: 0

Hypothesis testing

We employed a 3-step process to test the hypotheses that: (1) site types with lower management intensity (i.e. prairie and oldfield powerline corridors) would harbor more taxonomically and functionally diverse flowering plant communities; (2) more diverse flowering plant communities would lead to more diverse bee communities, and (3) more diverse bee communities would lead to greater seed set, but with functional diversity indices showing stronger patterns than taxonomic indices (Table 1).

Following Miles (2014), we first used the Variance Inflation Factor (VIF) in the same manner as described above to narrow our set of predictor variables. After removing bee variables with high collinearity, we were left with the two measures of bee abundance (honey bees and all bees combined), bee Shannon diversity, and the community-weighted mean for bee body size. After removing floral variables with high collinearity, we were left with floral unit abundance, flowering plant species Shannon diversity, flowering plant functional evenness, flowering plant functional dispersion, the community weighted mean for plant native status, and the community weighted mean for flower size (Fig. 2).

Second, we used the leave-one-out cross-validation method (LOOCV) to compare the null model to univariate models (either linear models or generalized linear models, depending on the nature of the response variable; see below details) with the predictor variables retained from step 1. Cross-validation is a resampling method that provides an error value indicating model fit; this approach usually returns results similar to AIC but is more appropriate for data with unusual distributions and reduces risk and error during model averaging (Zhang and Liu 2023).

As part of the second step, models that had a lower cross-validation error value than the null model were examined more closely. Predictor variables that performed better than the null model and also had statistically significant p-values ($p<0.05$) and confidence intervals that did not overlap zero were considered meaningful and were used for further analyses in Step 3. If only one model satisfied these requirements, we present the results of that model without proceeding to step 3.

If multiple univariate models satisfied the requirements of step 2, we proceeded to step 3 in which we created a model set including all possible combinations of the remaining predictor variables. Model sets were created using the dredge() function from the MuMIN package in R and compared against the null model using LOOCV. All models that performed better than the null in this round were model averaged (using weighted RMSE) to determine the effect of each predictor variable on the response variable. All inferences were based on those model-averaged parameter estimates. We utilized this 3-step process for comparing model

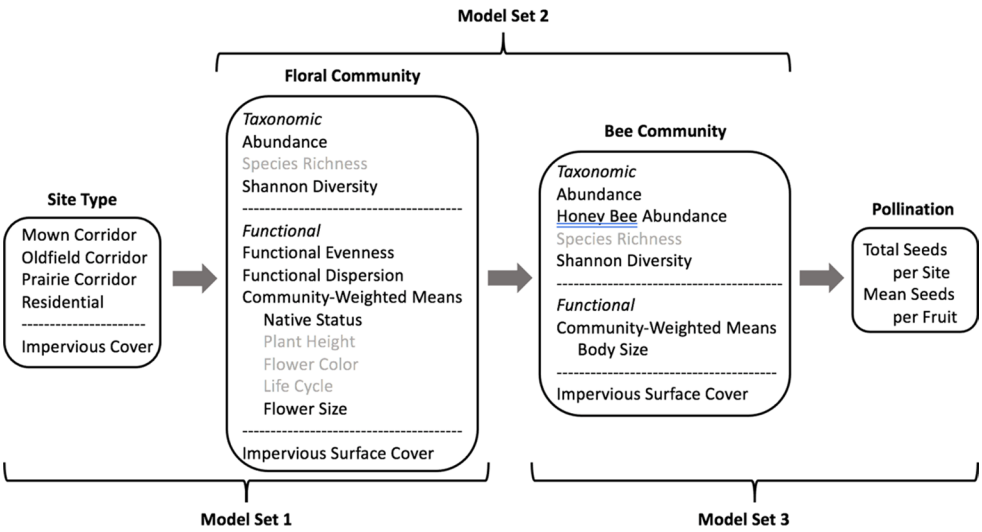

Fig. 2 Flow diagram of the analyses. Terms inside the boxes were measured and considered for initial analyses. Terms in light grey were excluded through VIF analysis. Results for Model Set 1 are found in Table 2, results for Model Set 2 are found in Table 3, and results for Model Set 3 are found in Table 4

fit to reduce the total number of models being compared and to reduce selection bias when determining our final models.

To understand the relationships between site type, flowering plants, bees, and pollination services in our study area, we ran a series of models (Fig. 2). The first set of models had flowering plant variables as the response variables, and both site type (mown corridor, oldfield corridor, prairie corridor, or residential neighborhood) and percent impervious surface cover as the predictor variables. We adjusted the error distribution family of our models or transformed our variables as needed to achieve models with residuals following a normal distribution. We log-transformed total number of flowers and flowering plant functional evenness, and square root-transformed the community-weighted mean of flower size, and used each of these response variables in separate linear models. Flowering plant functional dispersion and the community-weighted mean of flowering plant native status were each analyzed with generalized linear models with a quasibinomial distribution. Flowering plant Shannon diversity was not transformed and was analyzed with a linear model. To check for statistical artifacts due to our unbalanced study design (4 mown, 4 oldfield, 4 prairie, 12 residential), we ran a second version of this set of models where the values for all three residential neighborhoods in a cluster were averaged together to create a "representative" residential site. Results for this balanced version of the models (4 mown, 4 oldfield, 4 prairie, 4 residential) did not differ meaningfully from the original version of the models; comparison of model versions can be found in Appendix S4.

The second set of models included the flowering plant variables (untransformed) and percent impervious surface cover as predictor variables, and bee community variables as the response variables. To achieve models with residuals following a normal distribution, we log-transformed the total abundance of bees and bee Shannon diversity, and square root-transformed the total abundance of honey bees. The community-weighted mean for bee body size was not transformed and was analyzed using a linear model.

Finally, the third set of models included bee community variables and impervious surface cover as predictor variables, and seed set as the response variable. We ran linear models for the untransformed values for both the total number of cucumber seeds produced per site and the mean number of cucumber seeds produced per fruit per site.

Results

Summary statistics

We identified a total of 154 blooming plants (115 species plus 43 genera for which individual species could not be differentiated -- primarily ornamental plants in residential gardens) across all sites and sampling dates. Floral unit abundance ranged from 4,125 floral units in a prairie corridor to 260 floral units in a residential neighborhood in the same research cluster. Flowering plant species richness ranged from a high of 42 taxa in one residential neighborhood to a low of six taxa in two mown powerline corridors.

Flowering plant taxa with more than 2000 floral units counted across the study included *Erigeron sp.* (fleabane), *Lavandula angustifolia* (lavender), *Leucanthemum sp.* (daisy), *Plantago lanceolata* (ribwort plantain), *Trifolium pratense* (red clover), and *T. repens* (white clover). Flowering plant taxa present in ten or more research sites included *Daucus carota* (wild carrot), *Leucanthemum sp.*, *Oxalis sp.* (wood sorrel), and *T. repens*. Community-weighted means indicated that almost all research sites were dominated by non-native plants and plants with perennial life cycles. Half of our 24 research sites were dominated by plant species with white flowers.

The bee bowls captured a total of 1,947 individual bees from 41 species and 1,171 individuals from five morphospecies (including three morphospecies of *Lasioglossum*, three *Brachynomada* individuals, and one individual unidentified due to degradation during storage) across all study sites. The timed observations added 379 individual bees from ten species and six individual bees identified only to genus or family (mostly smaller bees or bees that moved too quickly to be identified to species). In combination, we recorded a total of 44 species and eleven morphospecies across all sites.

The most common bees from our samples were the three *Lasioglossum* morphospecies (1,167 total individuals combined). Other taxa that were well represented included *Halictus rubicundus* (690 individuals), *Augochlorella aurata* (269 individuals), *Ceratina mikmaqi* (234 individuals), *Halictus ligatus* (232 individuals), *Agapostemon virescens* (186 individuals), and *Apis mellifera* (155 individuals). The community-weighted means for bee body size fell between 8 and 11.5 mm for all sites. Although our sentinel plants were a cucurbit species, we did not observe or collect any species of squash bees (genera *Peponapis* and *Xenoglossa*).

We collected 5,383 seeds from the 300 harvested cucumbers, for an overall average of 17.9 seeds per fruit. The maximum number of seeds in a fruit was 257, and 18.3% of collected fruits (55 total) contained no seeds.

Hypothesis testing

Both flowering plant Shannon diversity (model $R^2 = 0.356$) and the community-weighted mean (CWM) of flower size (model $R^2 = 0.221$) showed statistically significant responses to site type (Table 2). Flowering plant Shannon diversity was lower in mown corridors than in residential sites (Fig. 3A),

Table 2 Results of hypothesis testing for flowering plant community response variables. Statistically significant variables are shown in bold. Degrees of freedom = 19 for all models

Response Variable	Predictor Variable	Estimate	SE	t value	*p*-value	Model R^2
Floral Unit Total Abundance						
	Intercept (Site Type – Mown)	7.190	0.554	12.984	<0.001	0.140
	Site Type – Oldfield	-0.663	0.554	-1.197	0.246	
	Site Type – Prairie	0.200	0.549	0.363	0.720	
	Site Type – Residential	-0.295	0.444	-0.665	0.514	
	Impervious Surface Cover	0.003	0.011	0.297	0.770	
Flowering Plant Shannon Diversity						
	Intercept (Site Type – Mown)	0.984	0.349	2.819	0.011	0.355
	Site Type – Oldfield	0.509	0.349	1.460	0.161	
	Site Type – Prairie	**0.894**	**0.346**	**2.583**	**0.018**	
	Site Type -- Residential	**0.825**	**0.280**	**2.948**	**0.008**	
	Impervious Surface Cover	0.004	0.007	0.631	0.535	
Flowering Plant Functional Evenness						
	Intercept (Site Type – Mown)	-0.985	0.223	-4.412	<0.001	0.098
	Site Type – Oldfield	0.302	0.223	1.352	0.192	
	Site Type – Prairie	0.148	0.222	0.666	0.513	
	Site Type – Residential	0.200	0.179	1.116	0.278	
	Impervious Surface Cover	0.002	0.004	0.368	0.717	
Flowering Plant Functional Dispersion						
	Intercept (Site Type – Mown)	-1.293	0.277	-4.659	<0.001	0.183
	Site Type – Oldfield	0.517	0.269	1.919	0.070	
	Site Type – Prairie	0.312	0.271	1.149	0.265	
	Site Type – Residential	0.183	0.224	0.817	0.424	
	Impervious Surface Cover	0.005	0.005	0.985	0.337	
CWM Flowering Plant Native Status						
	Intercept (Site Type – Mown)	-19.265	6256.891	-0.003	0.998	0.713
	Site Type – Oldfield	-0.800	8758.233	0.000	1.000	
	Site Type – Prairie	21.343	6256.891	0.003	0.997	
	Site Type – Residential	-0.235	7171.200	0.000	1.000	
	Impervious Surface Cover	-0.070	0.039	-1.778	0.091	
CWM Flower Size						
	Intercept (Site Type – Mown)	1.678	0.445	3.773	0.001	0.221
	Site Type – Oldfield	0.512	0.445	1.151	0.264	
	Site Type – Prairie	0.583	0.441	1.323	0.202	
	Site Type – Residential	**0.778**	**0.357**	**2.180**	**0.042**	
	Impervious Surface Cover	-0.007	0.009	-0.763	0.455	

and the CWM of flower size also trended toward smaller flowers in mown powerline corridors (Fig. 3B). Impervious surface cover did not significantly affect any measures of the flowering plant community.

Only our models for bee abundance out-performed the null model and were statistically significant (Table 3). Bee abundance responded negatively to both impervious surface cover (model $R^2 = 0.217$, Fig. 4A) and flowering plant functional evenness (model $R^2 = 0.193$, Fig. 4B). Model averaging indicated that the two individual models were better than the combined model (Table 3). Both measures of seed set (total number of seeds and mean seeds per fruit) were negatively related to total bee abundance (Table 4; Fig. 5).

Discussion

Our study demonstrates that pollination services in suburban areas are affected by interactions between multiple pollinator taxa and their responses to site type. Contrary to our expectation, we found no relationship between bee functional traits and seed set, while total bee abundance had a negative effect on seed set. Total bee abundance was also negatively correlated with flowering plant functional evenness and percent impervious surface cover. Our results demonstrate that the habitat quality provided by powerline corridors varies with different management treatments, supporting the conclusions by Wojcik and Buchmann (2012), who found no clear positive or negative trends of powerline management on pollinators and pollination.

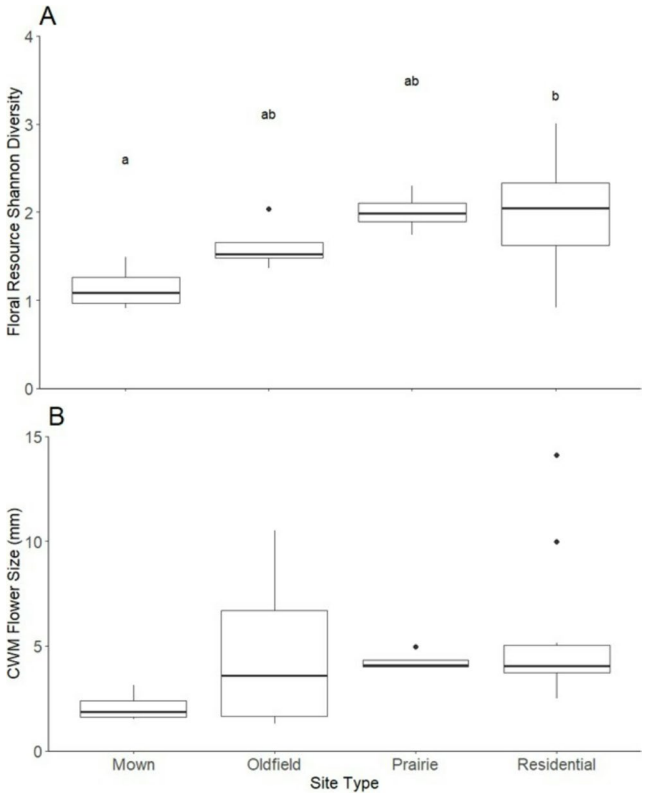

Fig. 3 Flowering plant community responses to site type. (**A**) Flowering plant Shannon diversity was higher in residential neighborhoods than in mown powerline corridors. (**B**) The community-weighted mean of flower size varied by site type (shown in Table 2), although pairwise comparisons did not reveal strong differences. Raw data were used to construct this figure, while models for this relationship were run using data transformed to fit model assumptions

Flowering plant responses to site type

Flowering plant Shannon diversity was lowest in mown powerline corridors, intermediate in oldfield and prairie corridors, and highest in residential neighborhoods (Fig. 3A). Neighborhoods showed greater within-type variation in Shannon diversity than any powerline corridor types. This is unsurprising given that previous studies have shown that flowering plant diversity can vary greatly across front yards and gardens (Burr et al. 2018; Lowenstein & Minor 2016, Martins et al. 2017), and previous studies of these sites showed residential neighborhoods have higher between-module (beta) diversity than powerline corridors (Hosler et al. 2025). Our results also indicated a trend toward mown powerline corridors having the smallest community-weighted mean flower size compared to the other three site types (Fig. 3B). The frequent intensive management in the mown powerline corridors may select for low-growing plants with smaller flowers able to avoid being disturbed during the growing season. Conversely, many ornamental plant varieties have been selectively bred for large, colorful blooms, and these ornamental plants are most likely driving the CWM trend in residential neighborhoods. However, it is unclear whether these large flowers are attractive to native bees or able to provide the quality and quantity of forage resources required by native pollinators.

Bee community responses to flowering plants

As expected, we found that some characteristics of the flowering plant community determined the composition of the bee community in our sites. However, some of our results seem counterintuitive at first: for instance, bee abundance responded negatively to flowering plant functional evenness (a functional diversity metric indicating the degree of evenness in representation of values across trait space), and pollination responded negatively to bee abundance. The negative relationship between bee abundance and flowering plant functional evenness is surprising, as much of the functional trait-based literature has shown positive relationships between functional diversity, biodiversity, and ecosystem functions. However, previous research suggests that our findings may be the result of strong urban environmental filtering on ecological communities (Ayers and Rehan 2021; Wilson & Jamieson 2019). Urban areas are likely to be dominated by weedy non-native or ornamental plant species with traits that allow them to survive urban stressors or have been selectively bred to be aesthetically pleasing (e.g., large flowers or tall height). Both situations could lead to low functional evenness. Comparing pairs of similar sites allows us to investigate the mechanisms leading to this pattern. For example, two mown powerline corridors, MGMC and CLMC, had the highest and lowest flowering plant FEve values (FEve values of 0.82 and 0.24, respectively). In the case of MGMC, high functional evenness was achieved because a very limited number of values for measured functional traits were present in the site, and the distribution of abundances of species with each trait value were relatively even (e.g. only three flower colors were detected in this site, but there were relatively similar abundances of all three flower colors). Conversely, CLMC included more values for the same trait (six color categories), but one trait value was vastly overrepresented (i.e. pink flowers were more than four times as abundant as the next most common flower color). If the dominant trait values (like pink flowers) in a functionally uneven plant community are attractive to many pollinators, we could see high bee abundance but poor pollination because bees favored foraging on these plants over our cucumber sentinel plants (which had yellow flowers). Conversely, if a functionally even plant community only includes a few values for a trait and all values are relatively similar in their abundance, there may not be enough "critical mass" of preferred trait values to attract many pollinators to

Table 3 Results of hypothesis testing for bee community response variables. Models shown here were all better than the null model according to the leave-one-out cross-validation (LOOCV) method. Models including statistically significant variables are shown in bold. Degrees of freedom = 22 for all models

Response Variable	Predictor Variable	Estimate	SE	t value/ z value	*p*-value	Model R^2
Bee Total Abundance						
Model 1	Intercept	-1.948	0.340	-5.734	<0.001	0.217
	Impervious Surface Cover	**-0.022**	**0.009**	**-2.469**	**0.022**	
Model 2	Intercept	-1.573	0.517	-3.046	0.006	0.193
	Floral Functional Evenness	**-2.335**	**1.017**	**-2.297**	**0.032**	
Model 3	Intercept	-3.094	0.263	-11.779	<0.001	0.121
	Floral Total Abundance	<0.001	<0.001	1.737	0.096	
Model 4	Intercept	-3.029	0.270	-11.228	<0.001	0.080
	CWM Floral Size	0.070	0.050	1.396	0.177	
Model 5	Intercept	-2.263	0.529	-4.280	<0.001	0.035
	Floral Shannon Diversity	-0.255	0.285	-0.839	0.381	
Full Model-Averaged Coefficients	Intercept	-1.462	0.647	2.189	0.029	
	Impervious Surface Cover	-0.015	0.013	1.156	0.248	
	Floral Functional Evenness	-1.529	1.333	1.124	0.261	
Honey Bee Abundance						
Model 1	Intercept	0.196	0.054	3.659	0.001	0.149
	Floral Shannon Diversity	-0.057	0.029	-1.961	0.063	
Model 2	Intercept	<0.001	<0.001	1.735	0.097	0.163
	Floral Total Abundance	<0.001	<0.001	2.067	0.051	
Model 3	Intercept	0.162	0.060	2.699	0.013	0.057
	Floral Functional Evenness	-0.137	0.118	-1.157	0.260	
Bee Shannon Diversity						
Model 1	Intercept	<0.001	<0.001	16.78	<0.001	0.130
	Floral Total Abundance	<0.001	0.001	1.81	0.084	
Model 2	Intercept	1.958	0.065	30.098	<0.001	0.031
	CWM Floral Native Status	-0.190	0.225	-0.845	0.407	
CWM Bee Body Size						
Model 1	Intercept	<0.001	<0.001	49.110	<0.001	0.153
	Floral Total Abundance	<0.001	<0.001	1.994	0.059	
Model 2	Intercept	2.481	0.048	51.347	<0.001	0.129
	CWM Floral Size	-0.016	0.009	-1.808	0.084	

the area. These results suggest that using a single metric of functional diversity like functional evenness may not be sufficient to characterize a community, and that using multiple functional diversity metrics or metrics that consider more dimensions of a community may be necessary to achieve a well-rounded perspective on ecological functions.

Total bee abundance in our study was also negatively related to percent impervious surface cover. Previous urban pollinator research has shown that impervious surfaces can act as a strong filter on bee communities, especially limiting the nesting sites available to ground-nesting species (Geslin et al. 2016) and limiting the amount of foraging patches (Jha & Kremen 2013). We anticipate that the multiple strong environmental filters in our study area selected for specific bee communities that drove pollination of our cucumber sentinel plants (Silva et al. 2020). These filters may include the presence of non-native plant species (Ayers and Rehan 2021), urban heat islands (Harrison and Winfree 2015), and the fragmented nature of urban habitats (Aguilar et al. 2006; Hadley and Betts 2011).

Bee community contributions to seed set

Previous studies have demonstrated either bee abundance (Steffan-Dewenter and Tscharntke 1999) or density (Blaauw and Isaacs 2014) to be positive predictors of fruit and seed set. However, our findings indicate that raw abundance may not be an indicator of pollination quality, as different bee species forage in different ways. For example, honey bee abundance may increase in more urbanized or disturbed areas (Theodoru et al. 2020, Winfree et al. 2011). The foraging behaviors of honey bees and other highly abundant generalist species' (e.g. *Halictus rubicundus*, *Lasioglossum fuscipenne*) may actually be detrimental to pollination quality in certain scenarios. Honey bees have been shown to forage intensively on individual plants (Diller et al. 2022), which may reduce pollen exchange between plants. Additionally, when a bee species exhibits generalist pollination behavior, the likelihood of successful conspecific pollination will decline with an increasing number of plant species

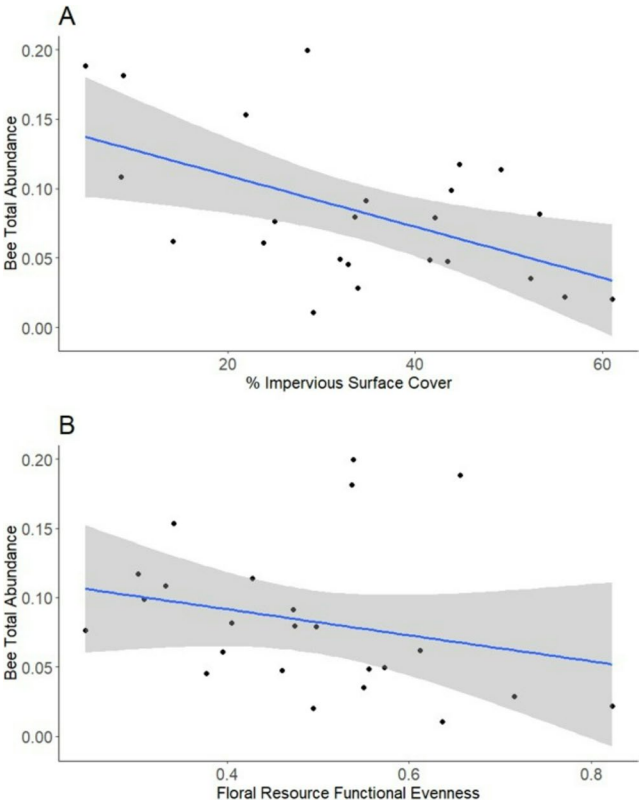

Fig. 4 Bee total abundance responded negatively to (**A**) impervious surface cover and (**B**) flowering plant functional evenness. Raw data were used to construct this figure, while models for this relationship were run using data transformed to fit model assumptions

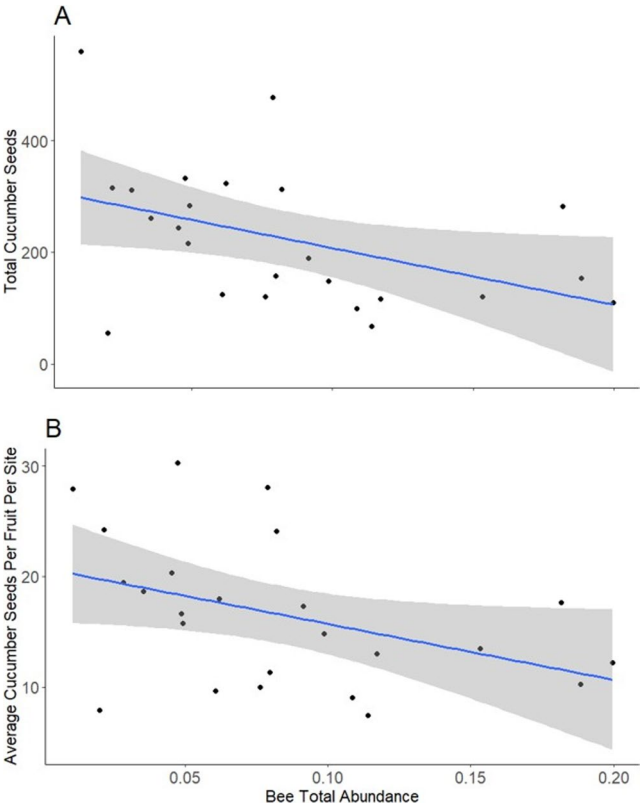

Fig. 5 Bee total abundance had negative effects on (**A**) the total number of cucumber seeds produced per site; and (**B**) the mean number of cucumber seeds produced per fruit per site. Raw data were used to construct this figure, while models for this relationship were run using data transformed to fit model assumptions

visited by that bee species (Burns et al. 2022). Our findings are similar to those of Leong et al. (2014), who found that bee abundance was higher in urban areas, yet urban areas also had lower seed set than their natural-area comparison sites, leading the authors to suggest that generalist bee species might be driving these patterns.

Limitations and future research directions

A notable limitation of our study stems from lack of agreement between researchers as to the selection and use of functional traits in analyses of ecosystem functions. We employed effect traits of both bees and flowering plants in our study. Bee traits like body size, hairiness, and tongue length were anticipated to directly impact pollination by affecting how bees interacted with flower reproductive parts and pollen, but body size was so closely linked to our other traits that we were unable to test the full dimensionality of bee functional diversity. Our flowering plant traits included flower size, flower color, and plant height, which were all anticipated to affect the attractiveness and accessibility of flowers to bees. However, our results were mixed, and we did not identify any drivers of bee or flowering plant functional diversity. We expect this is because our sets of traits did not include an ideal combination of response and effect

Table 4 Results of hypothesis testing for Seed Set response variables. Models shown here were all better than the null model according to the leave-one-out cross-validation (LOOCV) method. Models including statistically significant variables are shown in bold. Degrees of freedom = 22 for all models

Response Variable	Predictor Variable	Estimate	SE	t value	*p*-value	Model R^2
Total Number of Seeds						
Model 1	Intercept	308.72	44.82	6.887	< 0.001	0.184
	Bee Total Abundance	**-1013.14**	**454.64**	**-2.228**	**0.036**	
Means Seeds per Fruit						
Model 1	Intercept	20.774	2.366	8.779	< 0.001	0.169
	Bee Total Abundance	**-50.676**	**24.001**	**-2.111**	**0.046**	

traits. Like many other researchers, we have stumbled into the Raunkiaeran Shortfall (Hortal et al. 2015), where a limited knowledge of functional traits and ecosystem functions leads us to make suboptimal decisions regarding which traits we include in our analyses. There are as yet no formal guidelines on how to potentially mix both response traits and effect traits in analyses regarding ecosystem functions. Additionally, as functional diversity metrics are calculated using abundance data, our limited sample size may have further clouded the utility of functional diversity in this study.

We also acknowledge the limited number of sampling events in our study. As cucumber plants flower in mid-July in our study region, we chose to focus our surveys on summer-flying bees, but we acknowledge that our samples would have included more species with different traits if we had sampled in the spring as well. We also chose to focus solely on bees, although we acknowledge that flies in the family Syrphidae and certain beetle species are also important pollinators in our study area. We recognize that honey bees exist on the landscape as both feral and managed colonies, so including them in our total bee abundance measures may have skewed our understanding of the local bee community. Additionally, while cucumber is a common sentinel plant in pollination studies, the identity of the sentinel plant species can select for pollinators with certain traits. We did not choose to sample the bees directly interacting with our cucumber plants, but future studies should keep in mind that the subset of pollinators interacting with sentinel plants may have different traits than the broader pollinator community. Given these limitations, our results concerning bee diversity must be interpreted with caution, and our sentinel plant results may not be generalizable beyond cucurbit species. Additional data and analyses for these study sites are presented in Hosler et al. (2025), Garfinkel et al. (2023), and Hosler et al. (in review), which includes use of a second sentinel plant species. We thus encourage further investigations of these patterns in urban areas.

Subsequent studies should continue to address relationships among different urban green spaces, their flowering plant communities, and the pollinator communities they harbor. For example, with so much variation within types of urban green spaces (especially residential yards), are there specific methods of management and characteristics of the plant community that will consistently promote pollinator communities with specific characteristics? Healthy pollinator communities that provide high-quality pollination services promote reproductive success of insect-pollinated plants. In our study, we investigated this relationship using cucumbers, a common garden plant. Future research should focus on a broader variety of non-selfing and other potentially pollen-limited plants that could benefit from urban pollination services, including plant species on green roofs and in urban parks. Such investigations and continued comparison of different urban green spaces can inform pollinator-friendly land management strategies. For example, it appears that the mown powerline corridors in our system do not provide quality habitat compared to other methods of powerline management; if mown corridors are managed intensively for specific (as of now unstated) reasons, we suggest the power company establish “islands” of more natural vegetation with high Shannon diversity of floral resources. In the prairie and oldfield corridors, the power company could seed more native plant species throughout to increase flowering species Shannon diversity. However, given the relatively weak relationships detected in our study, our management recommendations may be very specific to our study system or geographic area, and thus further research is warranted.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s11252-026-01953-1.

Acknowledgements We would like to acknowledge the staff at the UIC Greenhouse, especially M. Frazel, for their assistance with growing and caring for our cucumber plants and for providing space for our plants. We are also grateful to the undergraduate research assistants who worked on portions of this project and the numerous homeowners who allowed us to conduct research in their backyards. We thank the editor and two anonymous reviewers for their thoughtful feedback. This work was funded by ComEd (Exelon Corporation) and the UIC Biology Department Hadley Award. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the authors and do not necessarily reflect the views of ComEd or Exelon Corporation.

Author contributions Contribution of authors: All authors designed the project. S.C.H. and M.R. collected the data. S.C.H. performed analyses and wrote the manuscript. All authors contributed to editing the manuscript.

Data availability Data sets utilized for this research may be found at: https://doi.org/10.5061/dryad.g79cnp5vs.

Declarations

Competing interests The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Emily Minor reports financial support was provided by Commonwealth Edison Company.

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