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RESEARCH

Proportion of native plants is a key predictor of pollinator richness in urban greenspaces

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Abstract
Pollinator declines are caused by a multitude of factors including pollution, global warming, disease, urbanization, deforestation, and habitat loss. Given the global increase in urbanization, identifying ways to support pollinators in cities has become an important conservation priority. Here, we investigate the effect of urbanization on pollinator richness. Using > 100,000 iNaturalist observations collected between 2014 and 2025 across 129 urban greenspaces in Florida, U.S.A., we investigate how native and non-native angiosperm richness, along with greenspace-level characteristics, influence pollinator richness. Pollinator richness was positively associated with overall angiosperm richness, regardless of origin. However, a higher proportion of native angiosperm species was associated with increased pollinator richness, whereas a higher proportion of non-native species was associated with decreased pollinator richness. Among greenspace-level landscape characteristics, pollinator species richness had a significant positive relationship with percent of impervious surface cover ($\beta = 0.0044$, $p = 0.010$), percent tree cover ($\beta = 0.0047$, $p = 0.001$), and greenspace size ($\beta = 0.0282$, $p = 0.037$), but was not significantly associated with grass cover ($\beta = 0.0020$, $p = 0.391$) or percent water cover ($\beta = -0.00005$, $p = 0.985$). Our results suggest that greenspace managers should prioritize enhancing floral richness—especially of native species—to support greater pollinator richness. Increasing tree cover and maintaining large greenspaces within cities may also contribute positively to pollinator richness. Leveraging broad-scale participatory citizen science data provides an avenue to further monitor pollinator richness within urban greenspaces.

Keywords Citizen science · Community science · iNaturalist · Pollination · Pollinators urban ecology · Angiosperms

Introduction

Pollinating insects are essential for ecosystem stability and agricultural productivity (Rader et al. 2016) but are declining in abundance (Wagner et al. 2021; Cornelisse et al. 2025). Such declines can disrupt plant communities, leading to broader ecological instability and reduced resilience in both natural and human-modified landscapes (Ramos-Jiliberto et al. 2020; Kawahara et al. 2021; Murphy et al. 2022). Pollinator declines are often linked to anthropogenic disturbances, such as habitat loss and fragmentation (Vanbergen 2013; Harrison and Winfree 2015), making urbanization a potential threat to pollinator richness. Urbanization tends to reduce both species richness and abundance of pollinators (Liang et al. 2023). But the varied nature of urban environments allows for a diverse range of native and non-native plants to grow, potentially increasing the range of pollinator species and community compositions that can be supported in urban ecosystems (Lowe and Foltz-Sweat 2017).

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Despite the potential negative impacts of urbanization, urban greenspaces (e.g., parks, nature reserves) can offer a sanctuary for native plants and pollinators in an otherwise hostile environment (Mata et al. 2021; Lepczyk et al. 2017). Urban greenspaces present unique conservation challenges (Aronson et al. 2017), as they need to balance the needs of public use and conservation (Miguez et al. 2025). For example, unmanaged or private urban and suburban natural areas can facilitate the introduction of non-native plants into urban greenspaces, either unintentionally through natural spread or intentionally through the planting of ornamental plants. However, pollinators do not always exhibit strong preferences for native plants compared to non-native plants (Harrison and Winfree 2015). Where native plants provide limited floral resources such as nectar and pollen later in the growing season, late-flowering exotic species can help sustain native pollinator populations (Staab et al. 2020). Additionally, greenspace size has been documented to be tied with species richness in some pollinator species due to increased habitat heterogeneity and resource availability (Hennig and Ghazoul, Zaninotto et al. 2023). However, other pollinator groups, such as bumble bees (Hymenoptera), which are considered more urban-tolerant, show no association between their abundance or diversity and greenspace size (Ulrich and Sargent 2025). Because of discrepancies in pollinator response to the outcome of management decisions such as plant community richness (e.g., Blaauw and Isaacs 2014; Dylewski et al. 2020) or greenspace size (e.g., Ulrich and Sargent 2025), understanding the ecological drivers of pollinator richness across broad spatial scales remains an important knowledge gap.

Due to their relatively high biodiversity and proximity to populated areas, urban greenspaces provide the public with ample opportunities to engage in citizen science (also referred to as community science or participatory science). Since it is often expensive and time-consuming to gather broad-scale pollinator richness data in the field, citizen science data provides a potential alternative to estimate pollinator richness among many urban greenspaces. Such citizen science data have been used for studies of pollinator-friendly gardens (Anderson et al. 2020), monitoring of pollinator services (Birkin and Goulson 2015), and citizen science project methods and participant motivations themselves (Bloom and Crowder 2020). Data from one of the most successful citizen science platforms—iNaturalist—are increasingly used in biodiversity research (Mason et al. 2025) and provide an opportunity to further understand pollinator richness among urban greenspaces. While iNaturalist data relies on opportunistic data collection and is less structured than field data, it can provide a larger spatial scope to study ecological trends (Díaz-Calafat et al. 2024).

Our primary aim was to test how native and non-native angiosperm richness and greenspace-level landscape characteristics impact pollinator richness in urban greenspaces (Fig. 1). First, we tested the hypothesis that angiosperm richness in urban greenspaces increases pollinator richness and that this impact depends on plant origin (native vs. non-native). Second, we examined how attributes of urban greenspaces (i.e., grass cover, impervious surface cover, tree cover, water cover, and greenspace size) relate to pollinator richness. Our work aims to provide evidence supporting the usefulness of participatory citizen science data for management of pollinator richness in urban ecosystems.

Methods

iNaturalist data and pollinators

iNaturalist is a citizen science platform accessible through a website (inaturalist.org) or application (iNaturalist) that allows users across the globe to upload visual or auditory observations of organisms. Observations are submitted for community review, where other users help identify recorded organisms. Observations are deemed “Research Grade”, when it contains complete metadata (e.g., date, location, media evidence) following the Data Quality Assessment (iNatHelp 2024) and have more than two-thirds agreement on identification at the species level. We obtained iNaturalist data using the iNaturalist data export tool on June 27th, 2025. We used this method because it provides detailed taxonomic resolution for each species, allowing us to filter observations by Superfamily and Subfamily. The original dataset was made up of 1,772,634 Research Grade iNaturalist observations of pollinators and angiosperms in Florida, U.S.A. Each observation in our dataset had an associated scientific name, date of observation, and location (latitude and longitude). Pollinators were defined as bees and apoid wasps (Superfamily Apoidea), bee flies (Family Bombyliidae), fruit and flower chafers (Subfamily Cetoniinae), butterflies and moths (Order Lepidoptera), and flower longhorn beetles (Subfamily Lepturinae). Angiosperms were defined as flowering plants from the subphylum Angiospermae. In addition, we obtained a list of non-native angiosperm species from iNaturalist by downloading species information via the API and filtering for “introduced” establishment means (iNatHelp 2025). We joined this list to our main dataset by scientific name to assign native and non-native status to each angiosperm species. We used iNaturalist observations from January 1, 2014, when the iNaturalist data is most reliable (Jacobs and Zipf 2017), to June 26, 2025, when the data was downloaded.

Study area and site selection

Florida, located in the southeastern United States, is a coastal, humid-subtropical state with a population of over 21.5 million people (United States Census Bureau 2020). It is one of the nation's fastest growing states, with large urban developments along its coastlines and interior (U.S. Census Bureau 2024). Our study areas are "urban greenspaces", defined as nature preserves, multipurpose greenspaces, nature trails, and other natural areas used by people within the urban system. We obtained greenspace boundaries from ParkServe (Trust for Public Land 2025), a nationwide dataset of parks and natural areas in the U.S. To isolate urban greenspaces, we filtered the ParkServe data to include only those located within urban areas as defined by the U.S. Census Bureau (2023) for Florida. This filtering resulted in a final dataset of 3,072 urban greenspaces. We restricted our original iNaturalist dataset to only observations in greenspaces ($N = 182{,}718$). We then removed all greenspaces which had less than 50 observations of either pollinators or angiosperms, leaving a final dataset of 106,171 observations within 129 urban greenspaces (Fig. 1). Qualitative exploration of observation cutoffs showed that 50 observations for both pollinators and angiosperms remove the high variability in species richness values in parks with fewer observations while maintaining a meaningful sample size (Table A2 – A3).

Park-level variables

We used remote sensing data from Google Earth Engine (Gorelick et al. 2017) to derive habitat characteristics within each urban greenspace. To represent land use and human influence, we calculated percent impervious surface cover, grass, trees, and water using the Dynamic World land use and cover datasets (10-meter resolution; Brown et al. 2022). While the Dynamic World data set contains nine landcover types (other habitats include: flooded vegetation, snow and ice, bare, crops, and scrub and shrub), we selected these four as we expected them to have the greatest influence on pollinator richness and because other habitat categories were rare in urban greenspaces, defined as occurring in less than half of the urban greenspaces examined (see Table A4). Within each greenspace polygon, we used a pixel-based histogram reducer to count the number of 100 m² landcover pixels

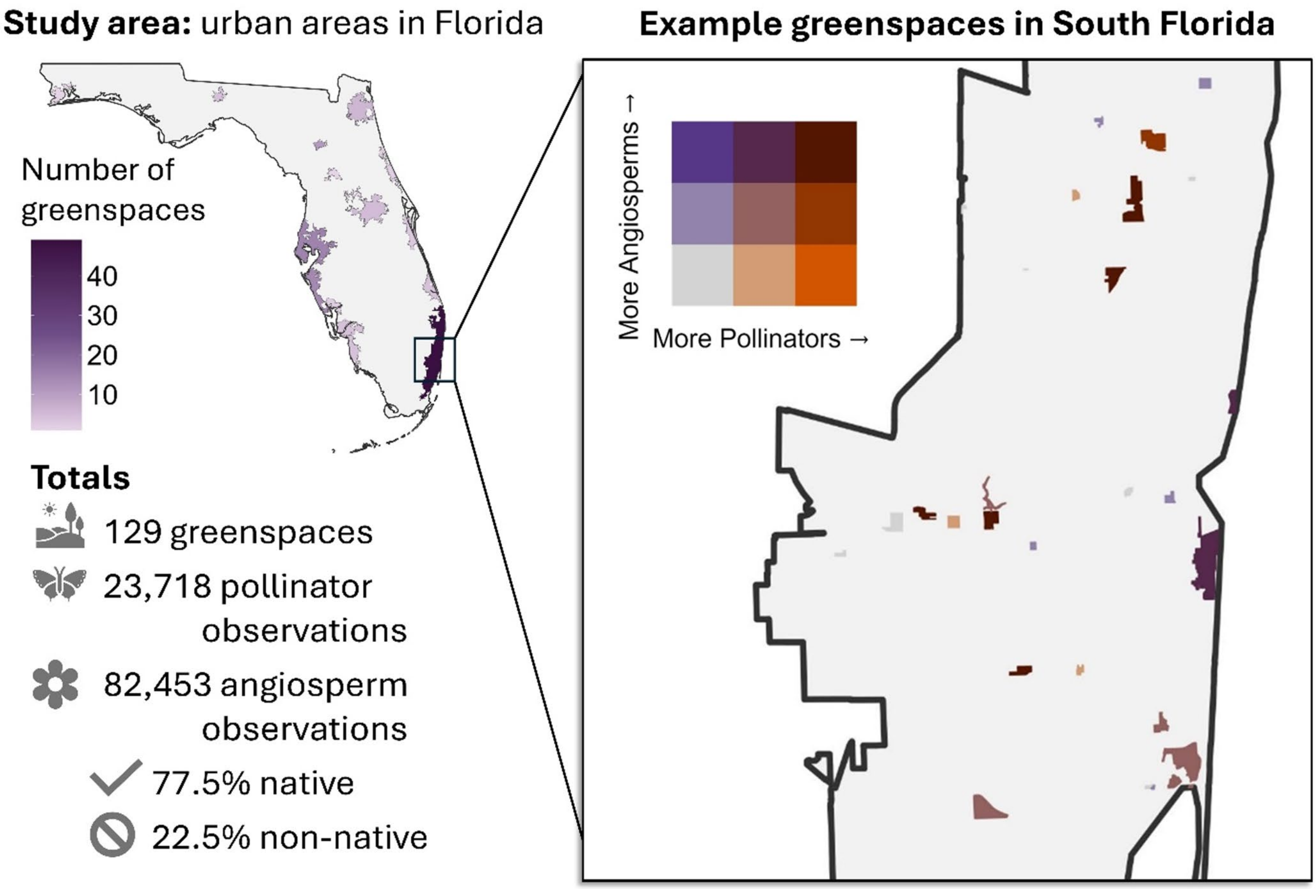

Fig. 1 Diagram illustrating the study area and citizen science data availability. The map of urban areas in Florida includes only those regions containing at least one greenspace that met our filtering criteria (≥50 pollinator and angiosperm observations). The example map of greenspaces in South Florida highlights a subset of the data to demonstrate the spatial resolution and structure of the data

belonging to each habitat type and calculated the percentage of each type relative to the total pixels in that polygon.

Statistical analysis

Our primary response variable was species richness, either pollinator richness or angiosperm richness measured as the number of unique species observed within each urban greenspace. While we acknowledge that iNaturalist data will not represent the true species richness values (i.e., absolute value), we assume that the data capture relative differences in species richness among urban greenspaces allowing us to infer true patterns. The ability of presence-only citizen science data to capture species richness patterns has been previously shown (Callaghan et al. 2020; Roberts et al. 2022). We additionally account for sampling effort differences in our model structure to ensure sampling differences among urban greenspaces are accounted for (see below).

To model the relationship between pollinator and angiosperm richness, we used generalized additive models (GAMs) using the mgcv package in R (Wood 2017), which allowed us to include a thin-plate regression spline for latitude and longitude to control for spatial structure (Wood 2003) and account for varying patterns throughout the state of Florida that may be due to variables not of interest (e.g., known species richness trends with latitude). To do this, we fit two GAMs. The first GAM included angiosperm richness as the response variable, and the log-transformed number of angiosperm observations, to address skewness and to account for uneven sampling effort, and spatial smooth (latitude and longitude) as predictors. We used this model to predict angiosperm richness after controlling for number of observations and spatial variation, using the predict.gam function from the mgcv package (Wood 2017). The second GAM included pollinator richness as the response variable with the predicted angiosperm richness from the first model, the log-transformed number of pollinator observations, and a spatial smooth as predictor variables. Both models used a negative binomial error distribution to account for overdispersion and were fit using the REML method (Wood 2011). Log-transformed sampling effort and predicted angiosperm richness were included as linear terms, while latitude and longitude were modeled as a thin-plate spline with a basis dimension (k) set to 50; selected to balance flexibility and reduce model complexity. Prior to modeling, we assessed multicollinearity and found no concerning correlations among predictors. We also evaluated model fit using the gam.check() function from the mgcv package to examine residual distributions and the relationship between response versus fitted values. To assess how much angiosperm richness improved model fit, we used Akaike Information Criterion (AIC) values to compare the resulting model to the null model. To evaluate whether native versus non-native angiosperm richness differentially influenced pollinator richness, we repeated the analysis separately for native and non-native angiosperm richness. In each case, we restricted the dataset to greenspaces with more than 50 angiosperm observations from the relevant group to ensure sufficient sampling coverage. We found that this threshold provided the best balance between sample size and statistical inference. This resulted in a sample size of 123 greenspaces for the native angiosperm models, and 104 greenspaces for the non-native angiosperm models. We repeated the analysis with proportion of native angiosperm richness and number of pollinator observations as predictor variables with pollinator richness as the response variable ($N = 123$). Further, to assess the importance of individual pollinator groups (defined as Superfamily Apoidea, Family Bombyliidae, Subfamily Cetoniinae, Order Lepidoptera, Subfamily Lepturinae, butterfly species [family Hesperiidae, Papilionidae, Pieridae, Lycaenidae, Riodinidae, and Nymphalidae], and moth species [Lepidoptera and not family Hesperiidae, Papilionidae, Pieridae, Lycaenidae, Riodinidae, and Nymphalidae]) on the observed trends, we repeated the described analyses using species richness calculated for each group that had at least 50 observations in multiple greenspaces.

To examine the influence of greenspace-level covariates on pollinator richness, we fit GAMs with a negative binomial family. We modeled pollinator richness as the response variable and percent grass cover, percent impervious surface cover, percent tree cover, percent water cover, and greenspace size for each greenspace as linear predictor variables, along with the log-transformed number of pollinator observations to control for sampling effort. We additionally modeled latitude and longitude using a thin-plate spline with a basis dimension (k) set to 50 to control for geographical variation in pollinator richness. Greenspace size was log-transformed due to its highly skewed distribution. We first ran a full model including all predictor variables, sampling effort, and geographic location. We then fit a null model including only the sampling effort and geographic location, followed by a set of five models, each including one of the park-level covariates, sampling effort, and geographic location. For all models, we evaluated model fit using the gam.check() function from the mgcv package to examine residual distributions and the relationship between response versus fitted values and found good model fit. We visualized the predicted relationship from each single-variable model and compared them using AIC values. To determine pollinator group-level responses to environmental covariates, we repeated the described analyses using species richness calculated for each group. For group-level analyses, the basis dimension for latitude and longitude was adjusted to sample size – 1 for groups with a sample size of less than 50.

Data analysis and availability

All data analysis was conducted in R (R Core Team 2025). All data and code are openly available on Zenodo: https://doi.org/10.5281/zenodo.17517393.

Results

Dataset summary

Our dataset spans over 10 years, from January 1, 2014, to June 26, 2025. Due to the exponential increase in user uploaded iNaturalist observations, most observations came from recent years (median = September 25, 2022; SD = 751 days). Of the 106,171 observations, 82,453 (77.7%) were angiosperms and 23,718 (22.3%) were pollinators. These included 773 unique pollinator species and 2,008 unique angiosperm species. Pollinator richness per greenspace ranged from 13 to 220 species (mean = 48.5 ± 29.0 SD), while angiosperm richness ranged from 5 to 448 species per greenspace (mean = 68.9 ± 58.1 SD). Of the angiosperms, 63,882 were native (77.5%) and 18,571 were non-native (22.5%). Of the pollinators, most were in the order Lepidoptera (observations = 20,619, species = 648), followed by superfamily Apoidea (observations = 2,635, species = 87), family Bombyliidae (observations = 248, species = 25), subfamily Cetoniinae (observations = 182, species = 7), and subfamily Lepturinae (observations = 34, species = 6) (see Table A1 for subfamily-level composition of dataset). In Lepidoptera, most observations were from butterflies (*N* = 15,515 vs. 5,104 for moths), whereas most species belonged to moths (*N* = 528 vs. 120 for butterflies). There were 129 urban greenspaces that met our filtering criteria, which encompass approximately 100,641 ha (389 mi²). Our study area represents 0.44% of Florida urban regions (87,958 mi²). For the group-level analysis, we found that only Lepidoptera and Apoidea had enough data to calculate species richness in parks, with sample sizes of 113 and 13 parks, respectively. For the subgroups, we had a sample size of 80 parks for butterflies and 28 parks for moths.

Relationship between angiosperm and pollinator species richness

Angiosperm richness was positively associated with pollinator richness and models including it fit well (adjusted $R^2 = 0.826$; deviance explained = 81.6%), though much of this fit was due to sampling effort and spatial structure alone (adjusted $R^2 = 0.782$; deviance explained = 79.1%). Nonetheless, including angiosperm richness improved model performance ($\Delta AIC = 7.84$), and this variable was a significant predictor of pollinator richness (Estimate = 0.001 ± 0.0003 SE, $P < 0.001$; Fig. 2). Sampling effort also had a strong positive effect on both angiosperm richness (Estimate = 0.52 ± 0.02 SE, $P < 0.001$) and pollinator richness (Estimate = 0.48 ± 0.03 SE, $P < 0.001$; Figure S1).

The proportion of native angiosperm species in a greenspace was positively related to pollinator richness with models indicating that a 10-percentage-point increase in the proportion of native plant species within a park is associated with an approximate 20% increase in expected pollinator richness. (Estimate = 1.83 ± 0.48 SE, $P < 0.001$; Fig. 2B). This model fit the data well (adjusted $R^2 = 0.826$; deviance explained = 82.1%), but much of this fit was attributable to sampling effort and spatial structure (adjusted $R^2 = 0.782$; deviance explained = 79.1%); however, the model including

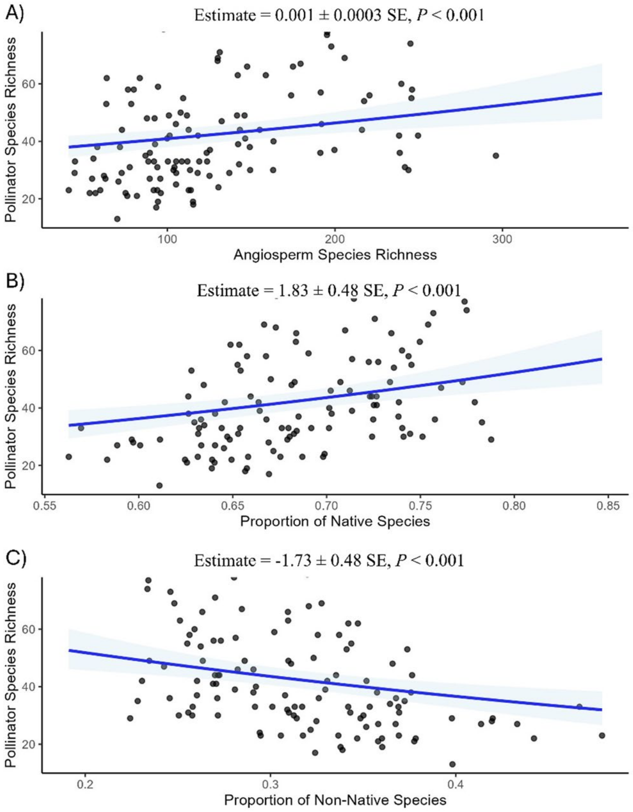

Fig. 2 Scatterplots comparing pollinator species richness with the richness of (**A**) all angiosperms, (**B**) the proportion of native angiosperm species, and (**C**) the proportion of non-native angiosperms across greenspaces. Each point represents an individual greenspace. Both pollinator and angiosperm species richness values are adjusted for sampling effort and spatial variation using Generalized Additive Models (GAM). The blue line shows the modeled relationship from the GAM model with 95% confidence interval. The estimated pollinator richness coefficient, standard error (SE), and associated p-value are reported for each model. Outliers were removed for display purposes; the full plot including all points is shown in Figure S2

the proportion of native angiosperms outperformed the null model (ΔAIC = 9.53), indicating that the proportion of native plants in a park adds explanatory power over spatial effects and sampling alone.

We additionally found a positive relationship between the native angiosperm richness and pollinator richness (Estimate = 0.001 ± 0.0004 SE, $P = 0.002$). This model also fit the data well (adjusted $R^2 = 0.822$; deviance explained = 80.7%), although much of the variation was explained by sampling effort and spatial structure alone (adjusted $R^2 = 0.790$; deviance explained = 79.1%). Nevertheless, including native angiosperm richness improved model performance (ΔAIC = 7.20), and it was a significant predictor of pollinator richness. Sampling effort remained a strong positive predictor of both native angiosperm richness (Estimate = 0.52 ± 0.02 SE, $P < 0.001$) and pollinator richness (Estimate = 0.50 ± 0.03 SE, $P < 0.001$).

The proportion of non-native angiosperm species had a significant, negative relationship with pollinator richness with models indicating that a 10-percentage-point increase in the proportion of non-native plant species within a park is associated with an expected 16% decrease in pollinator richness (Estimate = −1.73 ± 0.48 SE, $P < 0.001$; Fig. 2C), and the model fit the data well (adjusted $R^2 = 0.815$; deviance explained = 81.8%). Much of this fit was attributable to sampling effort and spatial structure (adjusted $R^2 = 0.782$; deviance explained = 79.1%); however, the model including the proportion of non-native angiosperms outperformed the null model (ΔAIC = 8.00), indicating additional explanatory power.

While a higher proportion of non-native species was associated with decreased pollinator richness, the opposite was true for raw non-native richness which had a weak positive association with pollinator richness. Non-native angiosperm richness also demonstrated good fit (adjusted $R^2 = 0.791$; deviance explained = 81.0%), though much of this fit was also explained by sampling effort and spatial structure alone (adjusted $R^2 = 0.775$; deviance explained = 80.3%). The model including non-native angiosperm richness showed only slight improvement compared to the null model (ΔAIC = 1.87), with a model weight of 0.69. Non-native angiosperm richness showed moderate evidence of a significant positive association with pollinator richness (Estimate = 0.003 ± 0.001 SE, $P = 0.045$; standardized estimate = 0.11). Sampling effort continued to be a strong predictor of non-native angiosperm richness (Estimate = 0.53 ± 0.03 SE, $P < 0.001$) and pollinator richness (Estimate = 0.50 ± 0.03 SE, $P < 0.001$).

When we repeated this analysis for taxonomic groups (Figure S3 – S6, we found a significant positive trend between Lepidoptera species richness and angiosperm species richness (Estimate = 0.001 ± 0.0004 SE, $P = 0.004$) and proportion of native angiosperm species (Estimate = 1.90 ± 0.59 SE, $P = 0.001$). We found a significant negative trend between Lepidoptera species richness and proportion of non-native species (Estimate = −1.87 ± 0.50 SE, $P = 0.001$). However, within the butterfly and moth subgroups, none of the angiosperm metrics were significantly related to species richness. Similarly, no significant relationships were found for Apoidea species.

Greenspace-level covariates and pollinator species richness

In the full model which contained all predictor variables, we found a significant positive relationship between pollinator richness and percent impervious surface cover (Estimate = 0.004 ± 0.002 SE, $P = 0.010$), percent tree cover (Estimate = 0.005 ± 0.001 SE, $P = 0.001$), and greenspace size (Estimate = 0.028 ± 0.013 SE, $P = 0.037$; Fig. 3). We found no significant relationship between pollinator richness and percent grass cover (Estimate = 0.002 ± 0.002 SE, $P = 0.391$) or percent water cover (Estimate = −0.00005 ± 0.002 SE, $P = 0.985$).

The AIC comparisons indicated that the full model, which included all predictor variables, performed best (AIC = 980.28), compared to the second-best model, which included only percent tree cover (AIC = 982.30). The full model showed high explanatory power (adjusted $R^2 = 0.797$; deviance explained = 81.6%). However, much of this was attributable to sampling effort and spatial distribution, as the null model—including only these factors—also explained a large portion of the variance (adjusted $R^2 = 0.770$; deviance explained = 77.7%). Nevertheless, the full model significantly improved fit compared to the null model (ΔAIC = 13.29).

When we repeated this analysis taxonomic groups (Figure S7 – S10), we found a similarly positive relationship in Lepidoptera species richness and impervious surface cover (Estimate = 0.006 ± 0.002 SE, $P = 0.003$), tree cover (Estimate = 0.005 ± 0.002 SE, $P = 0.002$), and greenspace size (Estimate = 0.035 ± 0.032, $P = 0.023$), and a non-significant relationship with grass cover (Estimate = 0.002 ± 0.003 SE, $P = 0.561$) and water cover (Estimate = 0.0009 ± 0.003 SE, $P = 0.729$). However, no significant trends were observed for the butterfly and moth subgroups, nor for Apoidea, across any land cover variables.

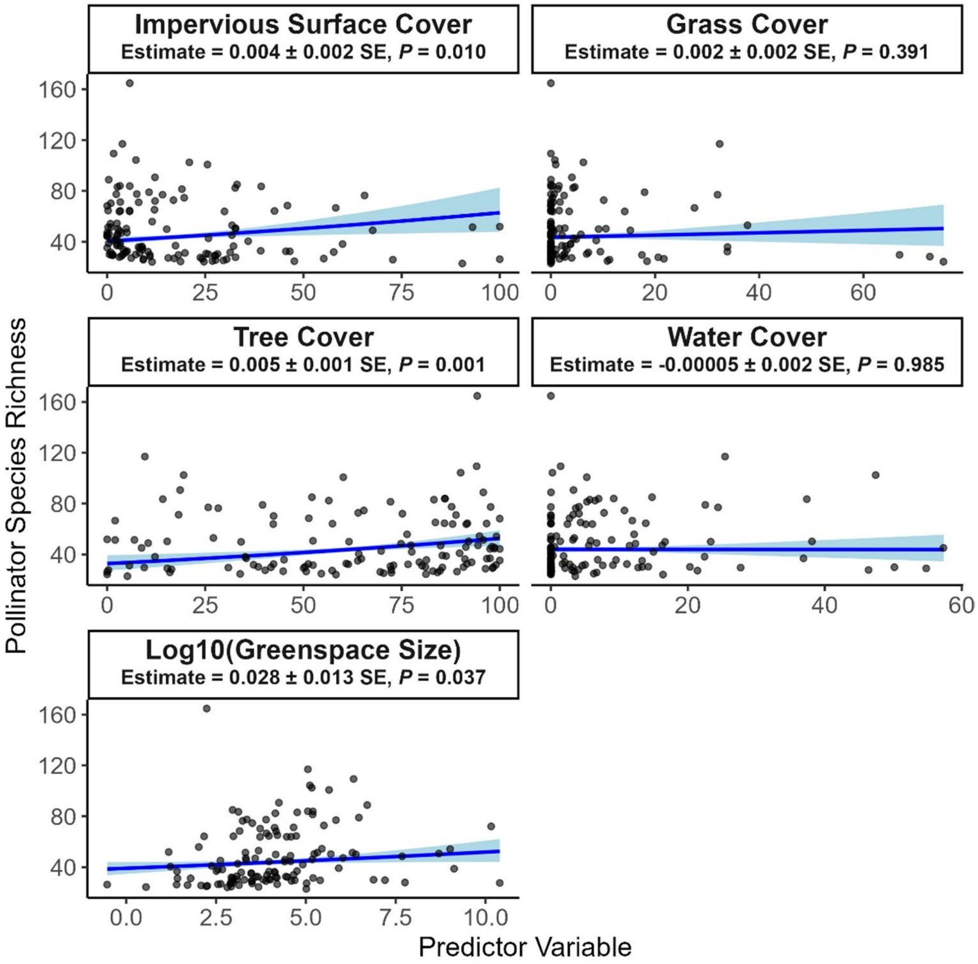

Fig. 3 Scatterplots illustrating the relationship between pollinator species richness in each greenspace and key environmental covariates: grass cover (%), impervious surface cover (%), tree cover (%), water cover (%), and log-transformed greenspace size (ha). Pollinator species richness values have been adjusted to account for variation in sampling effort. The blue line shows the modeled relationship between each environmental variable and adjusted pollinator richness, calculated using a Generalized Additive Model with a thin-plate regression spline for latitude and longitude. The shaded ribbon depicts 95% confidence intervals around the predictions. The estimated pollinator richness coefficient, standard error (SE), and associated p-value are reported for each model

Discussion

Our analysis of 129 urban greenspaces across Florida demonstrates that plant richness, and particularly native plant richness, is a key predictor of pollinator richness in urban environments. While pollinators in our study are mostly represented by butterflies and moths, models indicated that angiosperm richness, regardless of plant location of origin (i.e., native or non-native) increased pollinator richness, but there was clear evidence that parks with proportionally more non-native species have lower pollinator diversities. This is illustrated by looking at our "most native" and "most non-native" parks in the data set which models predicted should have ~100 pollinator species vs. <50 species (Fig. 2). Among environmental factors, tree cover was most strongly associated with pollinator richness, while greenspace size showed modest positive effects. Impervious surface cover also showed a positive relationship with pollinator richness, likely reflecting the benefits of moderate disturbance or a greater richness of habitats being sampled by iNaturalist users. Taken together, our results support existing conservation goals for pollinators which prioritize planting flowering plants (Morales and Traveset 2009; Kral-O'Brien et al. 2021) and support the important goal of maintaining large urban greenspaces with a high proportion of tree cover (Ulrich and Sargent 2025).

Non-native angiosperm richness was a weak but positive predictor of pollinator richness. This suggests that removing non-native species without concurrent increases in native plant richness could harm pollinator communities, at least in the short term. For example, some non-native perennials and even weedy species have been shown to support a wide range of pollinators, including during seasonal gaps in native floral availability (Lowenstein et al. 2019; Koyama et al. 2018; Frankie et al. 2019; Seitz et al. 2020). Non-native plant species may also support more generalist species interactions and network nestedness (Zaninotto et al. 2023). This suggests that management strategies should emphasize replacement rather than simple removal with a focus on maintaining richness while shifting communities toward native-dominated floras. Our results suggest that restoration should focus on parks that have relatively low plant richness where targeted plantings of native species could have outsized impacts on pollinators, because those additions represent a large proportional increase in native plants. This adds to an increasing body of evidence that native plants support pollinator richness (Morales and Traveset 2009; Fukase and Simons 2016; Kral-O'Brien et al. 2021) and that managing for diverse plant communities is paramount (Daniels et al. 2020; Watson et al. 2022). Together, these results highlight that managing pollinator richness requires balancing native versus non-native plantings. Diverse plant communities with a variety of foraging and pollination niches support a greater number of pollinator species (Frankie et al. 2019), and non-native gardens can increase those niches (e.g., Staab et al. 2020) albeit not as effectively as native plants.

Aside from plant richness, we found localized, within-greenspace characteristics were associated with increased pollinator richness, including tree cover, impervious surface cover, and greenspace size. The strong effect of tree cover aligns with prior work showing that forested areas provide resources such as nesting sites and floral availability for bees and other pollinators (Ulyshen et al. 2023; Mola et al. 2021a, b). In urban ecosystems where surrounding forests are often limited, tree cover within greenspaces may be especially important for supporting pollinator communities. We further speculate that forested parts of greenspaces may also provide bare ground nesting habitat (e.g., for bees) and larval habitat, particularly in larger greenspaces, leading to the positive effect size we found in our analysis (Fortuin and Gandhi 2021; Habel et al. 2022). The positive effect of impervious surface cover was surprising, given that many studies report negative associations with pollinator richness (Gerner 2020; Kaiser and Resasco 2024). However, urban pavements with their cracks, interstices, and break lines, were reported to provide novel habitats to many ground-dwelling insects, including wild bees and wasps (Weber et al. 2024). In addition, some pollinators, such as large-bodied bees, can be positively associated with impervious surfaces (Bennett and Lovell 2019). It is also possible that impervious surface cover reflects habitat heterogeneity, encompassing residential yards, gardens, or other mixed spaces that support pollinators (Larson et al. 2022; Liang et al. 2023; Kostro-Ambroziak et al. 2025). Pollinators may be able to benefit more from this habitat heterogeneity in larger parks where there remain sufficient patches of forested or natural vegetation. Further, parks with higher impervious surface cover may be less prone to pesticide use. Despite its limitations in capturing real-world heterogeneity, impervious surface cover measures are correlated with pollinator richness when compared to lower-impervious-coverage areas such as intensive agriculture (Wenzel et al. 2020) and shifts in pollinator functional richness favors some pollinator life histories (Ayers and Rehan 2021). This result, however, may also reflect sampling bias in citizen science data, as the amount of impervious surface cover in urban greenspaces may influence where observers can access and record pollinators within an urban greenspace. Finally, the positive effect of greenspace size is consistent with ecological theory and empirical evidence, likely reflecting greater habitat heterogeneity and foraging resources in larger parks (Zaninotto et al. 2023; Hennig and Ghazoul 2011).

In contrast to the above variables, we found no significant relationship between pollinator richness and water or grass cover. Water features may provide localized benefits for individual pollinators, but they were not a key predictor of pollinator richness across our dataset. This may be partially due to pesticide use, where higher water cover in certain parks could result in more pesticide applications to control mosquitos (Florida Department of Agriculture and Consumer Services 2025). Additionally, grassland cover—derived from the Dynamic World dataset—likely includes a mix of natural meadows and manicured lawns, with the latter dominating urban greenspaces. Because lawn management (e.g., mowing frequency, pesticide use, vegetation richness) strongly shapes ecological value to pollinators (Brittain et al. 2010; Lerman et al. 2023; Morrison and Bright 2025), and these attributes were not captured in our analysis, we speculate that this may be why we did not detect a significant association between grassland cover and pollinator richness.

Our pollinator dataset mainly consisted of Lepidoptera (83.8% of species and 86.9% of observations), and we were unable to find significant trends outside of this group, which is likely due to the small sample size. Such bias towards Lepidoptera has been documented previously (Di Cecco et al. 2021; Díaz-Calafat et al. 2024) and can be attributed to their relative ease of observation and photography, as well as their charismatic appeal (Fischer et al. 2021). Bias is also evident in which insect observations reach Research Grade on iNaturalist. Small insects are less frequently identified to species, likely due to poor photograph quality, the need for diagnostic characteristics such as genitalia, or the requirement of dissection or DNA analysis (Di Cecco et al. 2021; Díaz-Calafat et al. 2024). Although previous research by Segre et al. (2023) has shown that Lepidoptera richness is a poor proxy for overall pollinator richness, they found that pollinator groups exhibited similar correlations with flower richness. Thus, while our observed relationship between pollinator and angiosperm richness may be generalizable across less-represented pollinator groups, the specific environmental covariates driving these patterns may vary depending on the present taxa. Given the bias towards Lepidoptera and their reliance on host plants, future work could examine the relationship between Lepidoptera richness and host plant richness. Citizen scientists could focus efforts on additionally documenting host plants to make this research possible.

Our study is geographically (i.e., the entire state of Florida) and taxonomically broad, providing generalizable patterns by encompassing over 106,000 observations across 129 urban greenspaces throughout Florida. This was possible by leveraging broad-scale citizen science data from iNaturalist. However, there are biases and data limitations. Given our study's focus on urban greenspaces, where observation density is likely driven by accessibility and infrastructure, there may be some biases in which urban greenspaces, and to what extent, they are sampled (Geldmann et al. 2016). Although we did account for sampling effort in our analysis, we could not account for bias towards recording certain species of pollinators or angiosperms, but we expect such biases to be consistent across the urban greenspaces (i.e., participant-level biases would be consistent across Florida). To reduce structural biases in the data, and increase the usability of such citizen science data, park managers can include signage or posters at the entrance to greenspaces encouraging users of citizen science platforms such as iNaturalist to use dedicated citizen science projects when possible (Deacon et al. 2023). Increasing the availability and visibility of citizen science initiatives within parks can improve the quality and utility of biodiversity data (Meeus et al. 2023), allowing for a growing dataset to be collected over time, and potentially allowing for the monitoring of restoration success in the future by tracking changes in pollinator and floral richness. Future work could build on our broad-scale approach by incorporating trait-based plant characteristics—such as nectar production or floral morphology—that more directly link floral resources to pollinator foraging preferences (e.g., Zeng et al. 2023). Such fine-scale data would complement our richness-based analysis by providing insight into the functional relationships that structure urban pollinator communities.

Additionally, we did not explore landscape-level factors that affect pollinator richness. One such factor could be human population density, which has been shown to affect pollinator behavior (Persson et al. 2022). Another is pesticide application or drift, given that pesticides have been documented in pollen samples within Florida urban parks (Démares et al. 2022). Further, habitat connectivity has been described to be positively correlated with pollinator richness and abundance (Graffigna et al. 2024). Future research could explore how landscape-level factors, such as human population density, pesticide drift, or habitat connectivity, influence the relationship between pollinator and angiosperm richness.

Conclusion

Using a large, spatially explicit citizen science dataset, we found that angiosperm richness, greenspace size, and landscape management were key drivers of pollinator richness. Our results offer a clear takeaway for urban greenspace managers, suggesting they should consider (1) planting more flowers, preferably native species which will have a more reliable impact on increasing pollinator richness, and (2) maintaining a richness of habitats, including forested, grassland, and mixed-use areas that help maximize pollinator richness. Our results also illustrate how citizen science can help understand and track pollinators in urban greenspaces.

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Author contributions TM, JF, BMM, NP, and CTC conceptualized the research questions. TM, BMM, and CTC led the data analysis. TM and CTC drafted the manuscript. All authors contribued to reviewing and editing the manuscript.

Data availability All data and code are openly available on Zenodo: https://doi.org/10.5281/zenodo.17517393.

Declarations

Competing interests The authors declare no competing interests.

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