



Native-plant landscaping enhances floral visitation by insects and birds in residential yards

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Abstract

Native plant landscaping is gaining traction in cities worldwide for its potential to enhance biodiversity and ecosystem function. However, it remains unclear to what extent native plants increase animal-floral interactions compared to conventional landscaping such as lawns and ornamental plants. We quantified insect and bird flower visitation networks in 31 residential yards across Greater Los Angeles, California. We also assessed whether native plants in low-income areas, often lacking green amenities (the luxury effect), can support diverse floral-visitation networks. From April to September 2023, we conducted three visits each to 18 paired yards (one native-planted, one conventional) and 13 additional native-landscaped yards. We used timed area searches to quantify insect and bird floral visitation and collected predictor data on air quality, temperature, plant cover, species identity, flower color, phenology, as well as housing prices. Native-landscaped yards supported 4× higher floral-visitor abundance and 1.5× higher richness than conventional yards, largely driven by bees (Apidae) and hoverflies (Syrphidae), which were positively associated with native plant richness, flower-color diversity, and housing price. Several native plant species, including California buckwheat (*Eriogonum fasciculatum*) and white sage (*Salvia apiana*) anchored interaction networks. Moreover, native-landscaped yards contained 2.9× more interactions (140 vs. 48 edges) and exhibited higher network connectivity and redundancy. Floral visitors used native plants disproportionately, with native plants receiving 88% of visits despite representing 80% of availability. Landscaping with native plants can bolster urban insect and bird floral networks, particularly in low-income areas, thereby improving ecosystem functioning in the core of cities.

Keywords California · Flower · Los Angeles · Luxury effect · Network analysis · Pollination

Background

Following the Industrial Revolution and throughout the 20th century, the United States experienced sustained periods of rapid urbanization, marked by large-scale human migration from rural areas to city centers (Bretagnolle et al. 2015). Urbanization, which greatly expands the footprint and

density of cities (Grimm et al. 2008), typically transforms formerly natural ecosystems into a land cover dominated by a developed environment with no natural analogs (Radeloff et al. 2015). While past research has clearly documented the adverse effects of urbanization on biodiversity worldwide (e.g., Simkin et al. 2022), there has also been a surge of recent work highlighting the importance of cities for distinct flora and fauna (e.g., Spotswood et al. 2021). Thus, a key need for urban planners is to understand how to augment cities to promote biodiversity and core ecological processes (Lepczyk et al. 2017; Rega-Brodsky et al. 2022).

A defining feature of cities worldwide are dense cores surrounded by sprawling residential areas (Seto et al. 2011). Residential areas comprised of predominately single-family homes with large yards and manicured lawns are a hallmark of urban development in the U.S. and many countries across the globe (Behnisch et al. 2022). The modified and often similar nature of landscaping in residential yards typically

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provides little usable habitat for native wildlife (Smallwood and Wood 2023), which may lead to homogenized conditions (Pearse et al. 2018). Nevertheless, over the past decade, research has uncovered the importance of residential yards in harboring wildlife populations (Lerman et al. 2023).

An essential modification of residential yards that has gained popularity is landscaping with native plants (Gerner and Sargent 2022). Native plants provide a myriad of services to biodiversity in urban environments (Tartaglia and Aronson 2024). For example, native plants introduce increased cover, food sources, and breeding locations that benefit insects and birds (Narango et al. 2017). Further, native plants modify the local climate (Espeland and Kettenring 2018), sequester carbon (Gratani et al. 2016), provide aesthetic benefits to humans through diverse floral displays and fragrances, and support food webs and soil health that enhance the ecological character of cities (Martinson 2020; Smallwood and Wood 2023). Although urban environments are associated with reduced diversity of wildlife species, especially rare and endangered ones, the intentional design of urban greenspaces with native plants can positively impact urban biodiversity and function (Goddard et al. 2010a; Tartaglia and Aronson 2024).

One component of wildlife that is particularly affected by urban plant landscaping is pollinators (Hall et al. 2017; Baldock 2020). Animal pollinators, including insects, e.g., bees (Apidae), butterflies (Lepidoptera), hoverflies (Syrphidae), and birds, e.g., hummingbirds (Trochilidae), rely on diverse floral resources for sustenance (Liang et al. 2023). However, the proliferation of non-native plant species in urban residential landscapes often reduces floral diversity, which can cause a potential disconnect between nectar availability, flowering times, and pollinator activity (Herrmann et al. 2023). Pollinators play a crucial role in providing essential ecosystem services and are experiencing a rapid decline in the face of global change, especially due to urbanization (Wagner 2020). Therefore, cities are at risk of running a deficit in pollination potential, which may carry over to affect other services beneficial to people, e.g., fruit yields (Liang et al. 2023).

Numerous studies have focused on the positive effect native plant landscaping in urban areas can have on pollinators (Vilà et al. 2009; Pardee and Philpott 2014; Gerner and Sargent 2022). Still, at least three critical gaps in our understanding of pollinator populations within cities remain. First, numerous studies highlight how native and non-native plants serve as resources for pollinators (Burghardt et al. 2015; Seitz 2020; Anselmo et al. 2023; Sánchez and Lara 2024). However, direct comparisons between native- and conventionally landscaped residential yards dominated by non-native plants remain limited. Given that residential yards represent substantial urban green space globally

(Lerman et al. 2023) and there is a rising trend toward converting conventional landscaping to native plants (Los Angeles Department of Water and Power 2022), quantifying the ecological benefits of this shift is increasingly urgent. Second, the relationship between pollinator populations and income levels remains uncertain (Baldock et al. 2019; Reynolds et al. 2024; Lewthwaite et al. 2024). The luxury effect, which is a positive correlation between biodiversity and wealth, has been documented for birds, plants, and mammals (Hope et al. 2003; Leong et al. 2018; Li et al. 2019), but evidence for pollinators is mixed. While some studies find no luxury effect for insects (Lewthwaite et al. 2024), others report income-driven differences in pollinator diversity and abundance (Baldock et al. 2019; Reynolds et al. 2024). Determining whether native plant landscaping mitigates or reinforces these wealth-biodiversity patterns may reveal disparities in pollinator-driven ecosystem services in underserved communities. Third, most urban pollinator studies provide snapshots at single time points, lacking data on seasonal dynamics. Understanding how pollinator communities vary across seasons in relation to flowering phenology and environmental conditions (e.g., temperature, air quality) is essential for designing landscapes that support pollinators throughout their active periods (Geslin et al. 2013; Sánchez and Lara 2024).

To address these gaps, we designed a study to assess the significance of native-landscaped residential yards in Greater Los Angeles as habitats for insect and bird floral visitors across varying income levels. We use the term ‘floral visitors’ to refer to insects and birds observed visiting flowers. While most likely act as pollinators, our methods do not confirm pollen transfer. We had five objectives for our study.

1. We employed a paired-sampling methodology to compare the means of the abundance and richness of bird and insect floral interactions (hereafter: floral-visitor abundance and richness) and environmental variables in nine yards landscaped with native plants (hereafter: native yards) and nine yards landscaped primarily with manicured lawns, shrubs, and ornamental plants (hereafter: conventional yards). We hypothesized that floral-visitor abundance and richness and environmental features, such as plant richness, would be higher in native yards, aligning with findings from similar avian focused studies (Lerman and Warren 2011; Smallwood and Wood 2023).
2. We quantified the relationship between environmental predictors (native and non-native plant richness, flower color, temperature, air quality) and home characteristics (yard size, housing value) with floral-visitor abundance and richness. We expected floral-visitor abundance and

richness to positively correlate with native plant diversity, flower color variety, yard size, and housing value, and negatively with poor air quality, consistent with findings from urban pollinator and biodiversity studies (Fukase and Simons 2016; Baldock et al. 2019; Daniels et al. 2020; Yong 2024; Lilkendey et al. 2025; Plant et al. 2025).

3. We assessed temporal patterns of floral-visitor interactions. We predicted that floral-visitor abundance and richness would increase with native-plant flowering peaks and decrease with non-native plants, while being negatively related to poor air quality (Farré-Armengol et al. 2016; Cole et al. 2020; Lilkendey et al. 2025).
4. We conducted a floral-visitor network analysis to understand the potential differences in network complexity between native and conventional yards. We predicted greater network complexity in native landscaped yards due to the potentially higher plant richness attracting more floral visitors (Goddard et al. 2010a; Hartop et al. 2018; Daniels et al. 2020), as well as other studies that have suggested that native plants host more diverse insect and bird communities (Burghardt et al. 2009).
5. We evaluated floral-visitor use of flowers relative to floral availability in yards to quantify whether certain

plant genera were used disproportionately to their availability. We predicted that floral visitors would exhibit selective foraging patterns, with disproportionate use of certain plant genera, similar to bumblebees (*Bombus* spp.) in montane riparian habitats of the Sierra Nevada that exhibited greater use of specific floral species relative to their availability (Cole et al. 2020).

Methods

Sampling sites

To fulfil our objectives, we used a combination of paired-sampling and unpaired surveys of individual native landscaped yards throughout Greater Los Angeles, California, USA (Fig. 1). Each pair consisted of a native and a nearby conventional yard (average distance between pairs = 93.9 m, range 27.8–297.6 m). We selected study sites using two complementary approaches. First, we incorporated a subset of yards from an existing sampling design that examined the role of native landscaping on bird foraging behavior (Smallwood and Wood 2023). Because that design was concentrated in higher-income neighborhoods, we expanded site

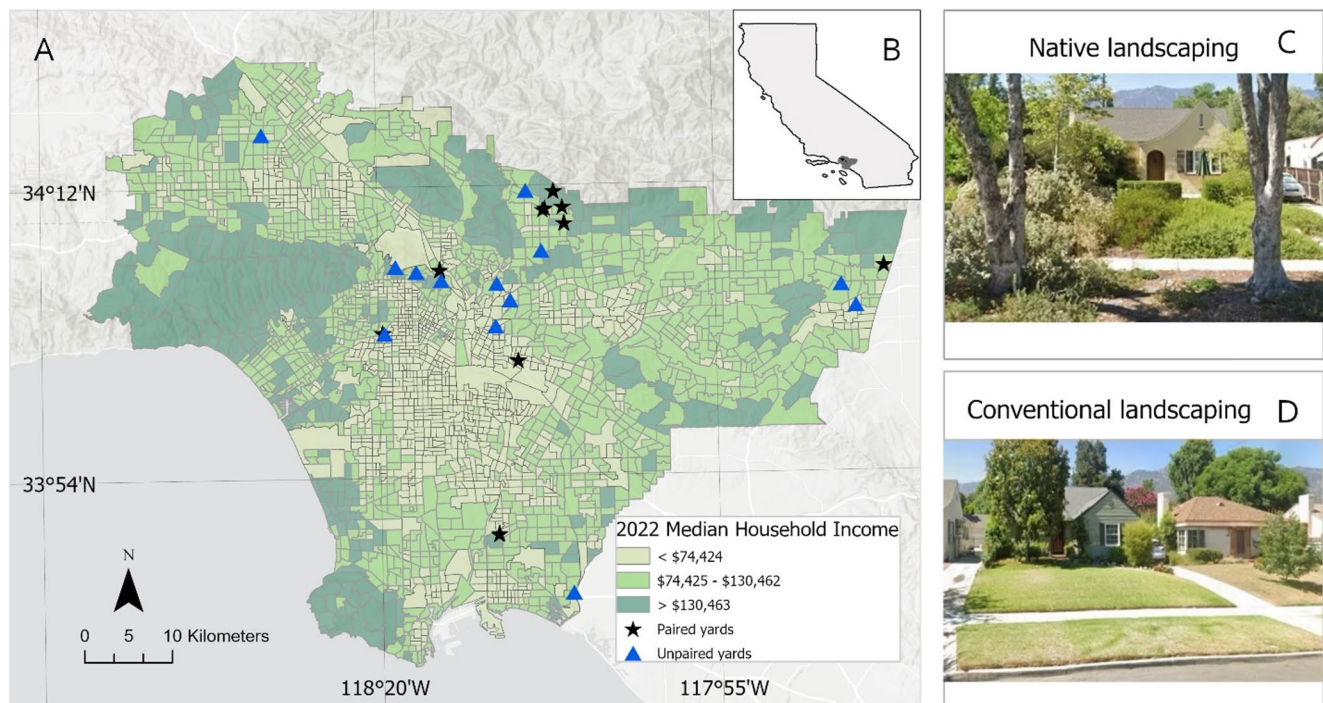


Fig. 1 Geographic distribution of sampling sites across the Greater Los Angeles region, with scale bar and north arrow shown created using ESRI ArcGIS software (Esri Inc., 2025). (A) Map of 31 residential yards included in the study, showing median household income by census tract (2022 U.S. Census). We classified median household income values into terciles (lower, middle, upper thirds) for visualization purposes only. Black stars indicate paired yards (n = 18 total;

9 pairs of native and conventional yards), and blue triangles indicate unpaired native yards (n = 13). Note that most paired yards overlap at the scale of this map. The inset (B) shows the location of the study region within California. Panels (C–D) show representative examples of (C) native-landscaped and (D) conventionally landscaped yards included in the study

selection to include lower-income areas to assess whether income disparities (i.e., the luxury effect) influenced environmental features that may affect floral-visitor activity. Using 2022 U.S. Census data we identified low-income census tracts suitable for study (U.S. Census 2022). We then walked selected neighborhoods, distributed flyers, and spoke directly with residents to recruit additional participants willing to allow research on their property. This approach yielded an additional 13 unpaired native yards, all located in lower-income areas, complementing the nine paired sites drawn from the original study. Homeowners provided information on landscaping practices, including pesticide use and plant composition. While most yards reported no pesticide use, a small number of homeowners indicated limited, targeted applications (e.g., localized treatments for ant infestations near tree bases). We did not quantify pesticide use in surrounding properties, which could influence floral-visitor communities at the yard scale. In total, we included 31 yards in the study: nine pairs of native and conventional yards ($n=18$) and 13 unpaired native yards distributed across the Greater Los Angeles region (Fig. 1).

Floral visitation and plant surveys

We used a timed-area search survey to observe floral-visitation by birds and insects from April to September 2023, to capture the shifting phenology of plants and their visitors from the spring to early fall (Watson 2003). Each yard was surveyed once in the functional spring-period for Southern California (April-May), once in the summer (June-July), and once in the fall (August-September). The duration of each survey was tailored to the yard size and availability of flowering resources. The average time per survey was 20 min (17 min for conventional yards; 23 min for native-landscaped yards). In general, larger yards with more flowering plants took longer to survey. Surveys focused on front yards to avoid variability in backyard sizes and landscaping practices (Locke et al. 2018). We conducted surveys in sunny and calm conditions between 10 a.m. and 4 p.m. to capture peak floral-visitation activity. Due to the geographic spread of sampling sites across the Greater Los Angeles region, we conducted surveys at one to two sampling sites per day and did not randomize survey order. Although this introduced some variation in time of day among sites, we restricted all surveys to periods of high floral-visitation activity.

In all study yards, we recorded the duration of each survey and documented all insect and bird floral interactions (Geslin et al. 2013). We walked slowly along a set path in each yard, examining every flowering plant and quantifying all insects and birds visiting flowers. For each interaction, we recorded the insect or bird species, and the plant species and its flower color. After observing one flowering plant, we

moved to the next until we surveyed the entire yard. On each visit, we also recorded animal-pollinated plant species that were not flowering to measure plant flowering phenology and assess potential shifting food availability throughout the season. Our method likely missed some floral interactions occurring in the yard while we were present. However, we opted for careful observation of each flowering plant to capture the full range of floral visitations with consistent effort across all plants.

For insects, we documented butterflies (Order: Lepidoptera), bees (Family: Apidae), flies (Order: Diptera), and wasps (Family: Vespidae). For birds, we focused on two hummingbird species that commonly visit flowers in our system: Anna's (*Calypte anna*) and Allen's Hummingbird (*Selasphorus sasin*). Since some homeowners did not permit sweep-netting, we relied solely on observation for data collection (Geslin et al. 2013). Given the observational nature of our surveys and the difficulty of identifying certain taxa (particularly bees) to species level in the field, we aggregated data at the family level for analyses. The first author (ER) completed all surveys to ensure consistency.

Yard size, air quality, temperature, and house price data

To determine the front yard area (size), we hand drew polygons using Google Earth Pro[®] to measure the boundaries of each yard and computed their respective areas (Google 2016). We recorded temperature and air quality data prior to each survey at the starting location of each walking route using the Apple Weather app (Apple[®] 2023) and the AirVisual app (IQAir[®] 2023). We note that both temperature and air quality data reflected regional patterns rather than yard-specific microclimates, as the apps provided neighborhood-scale rather than site-specific measurements. This approach allowed us to assess whether regional weather and air quality patterns correlated with insect and bird floral visitation across our study neighborhoods throughout the sampling period. We sourced home prices, used as a proxy for the luxury effect, from the Redfin real estate platform using their fall 2023 estimates for each property (Redfin 2018). Home prices ranged from \$583,700 to \$2,854,337 USD, with 12 homes priced below \$1 million, eight homes between \$1 million and \$1.5 million, and 11 homes above \$1.5 million. Our final environmental predictor variables included total plant richness, native plant richness, non-native plant richness, total flower-color richness, native flower-color richness, non-native flower-color richness, temperature, air quality, log(house price), and yard size.

Statistical analysis

We used R for all analyses (R Core Team 2025) and produced all graphics in R using “ggplot2” (Wickham 2016) and finalized each using Adobe Illustrator (Adobe Inc. 2019).

Objective 1 – Paired yard analysis

We quantified differences in floral-visitor abundance, floral-visitor richness, and environmental variables between paired native and conventional yards using generalized linear mixed models (GLMMs), with pair ID included as a random intercept to account for non-independence within yard pairs. We modeled floral-visitor abundance using negative binomial GLMMs to account for overdispersion in count data and included survey duration as an offset [$\log(\text{duration})$] to standardize counts by sampling effort. We modeled all richness variables (floral-visitor richness, plant richness, and flower-color richness) using Poisson GLMMs. We analyzed continuous environmental variables (yard size, temperature, air quality, house price) using Gaussian linear mixed models (LMMs) with pair ID as a random intercept. For house price, we log-transformed values to meet normality assumptions given the wide range across sites (\$583,700 to \$2,854,337).

We assessed model fit using DHARMA residual diagnostics to confirm appropriate variance structures and absence of overdispersion (Hartig 2025). We calculated within-pair standardized mean differences (Cohen’s d) as dimensionless effect sizes for display purposes, with 95% confidence intervals. We determined statistical significance from GLMM/LMM p -values using the R package “lmerTEST” (Kuznetsova et al. 2017), and we interpreted effect sizes as small ($|d| = 0.2$), medium ($|d| = 0.5$), or large ($|d| \geq 0.8$) following Cohen (Cohen 1988).

To assess differences in community composition between native and conventional yards, we conducted a Bray–Curtis dissimilarity analysis of floral-visitor communities using the vegan package in R (Oksanen et al. 2026). We visualized compositional differences using non-metric multidimensional scaling (NMDS) ordination and tested for differences between yard types using permutational multivariate analysis of variance (PERMANOVA; adonis2), with yard type (native vs. conventional) as a fixed effect and pair identity included as a permutation stratum to account for the paired sampling design. We additionally tested for differences in multivariate dispersion between yard types using a permutational test of homogeneity of dispersions (betadisper) (Oksanen et al. 2026).

Objective 2- Relationships between environmental features and floral visitors

To address our second objective, we quantified relationships between environmental predictor variables and floral-visitor abundance and richness using GLMMs with data from the nine paired yards only ($n = 18$). Given our modest sample size ($n = 9$ pairs) and the exploratory nature of this objective, we fitted separate univariate models for each of ten environmental predictors: total plant richness, native plant richness, non-native plant richness, total flower-color richness, native flower-color richness, non-native flower-color richness, temperature, air quality, $\log(\text{house price})$, and yard size. This approach allowed us to identify candidate predictors associated with floral-visitor abundance and richness while avoiding overfitting and model convergence issues common in small-sample mixed models. The paired sampling design, with pair ID as a random intercept controlled for neighborhood-level confounders (e.g., regional climate, surrounding habitat) that might otherwise create spurious associations between predictors.

We modeled floral-visitor abundance using negative binomial GLMMs (to accommodate overdispersion) and richness using Poisson GLMMs (no overdispersion detected). For each predictor, we compared linear and quadratic models using AIC, retaining quadratic terms when they improved model fit ($\Delta\text{AIC} > 2$). These included total flower-color richness, total plant richness, native plant richness, and native flower-color richness in floral-visitor abundance models, and total plant richness and native plant richness in floral-visitor richness models. We fitted all models using the “glmmTMB” package (Brooks et al. 2017) in R and assessed model fit using the “DHARMA” package (Hartig 2025).

Objective 3- Plant-floral-visitor interactions across seasons

To evaluate how floral-visitor abundance and richness varied across seasons between native-plant and conventionally landscaped yards, we used GLMMs with repeated measures. Unlike Objectives 1 and 2, which focused on spatially paired comparisons, we included all 31 study yards (22 native-plant landscaped, 9 conventionally landscaped) to maximize statistical power for detecting temporal patterns. We modeled floral-visitor abundance, flowering plant richness, native flowering plant richness, and non-native flowering plant richness using negative binomial GLMMs to account for overdispersion in count data, and floral-visitor richness using Poisson GLMMs. For continuous response variables (temperature and air quality), we used Gaussian linear mixed-effects models (LMMs).

All models included yard type (native vs. conventional) and sampling visit (spring, summer, fall) as fixed effects, along with their interaction (Type $\times$ Visit) to test whether yard type effects varied seasonally. We included yard location (Location) as a random intercept to account for repeated measurements within individual yards across the three visits. For abundance and richness models, we included survey duration as an offset term [$\log(\text{TimeSurveyed})$] to account for variation in observation time across surveys.

We evaluated main effects and interactions using Type III Wald χ^2 tests for GLMMs and Type III F -tests for LMMs. When significant effects were detected, we conducted pairwise comparisons using estimated marginal means with Bonferroni correction for multiple comparisons. We assessed model fit for GLMMs using DHARMA residual diagnostics to verify appropriate distributions and detect potential overdispersion or heteroscedasticity. All models were fitted using the “glmmTMB” and “lme4” (Bates et al. 2015) packages in R, with marginal means and post-hoc comparisons computed using the “emmeans” package (Lenth and Piaskowski 2025).

Objective 4- Floral-visitor networks

To evaluate plant–floral-visitor interactions between native and conventional yards, we constructed bipartite interaction networks using only the paired-yard subset (nine native yards paired with nine conventional yards). Plant species formed one set of nodes and floral-visitor families formed the other, and observed interactions represented edges. For each yard type, we pooled interactions across the nine paired yards to generate a native-yards network ($n=9$) and a conventional-yards network ($n=9$). We then conducted three comparative analyses:

1. *Paired-yard network comparison* - In the first analysis, we compared the overall structure of the native and conventional networks constructed from the paired-yard subset. This allowed us to quantify differences in realized floral interactions between yard types and to evaluate how network complexity is expected to change when a conventional yard is converted to a native landscape within a similar neighborhood context.
2. *Plant-origin effects* - In the second analysis, we evaluated how plant origin (native vs. non-native) influenced floral-visitation patterns while controlling for differences in plant-species richness between yard types. Because native yards contained more plant species than conventional yards, we used a bootstrapping procedure to subsample the pool of native plant species to match the number of non-native plant species observed across the paired conventional yards (10 species per iteration).

For each of 1,000 iterations, we randomly sampled 10 native plant species (with replacement) to rebuild a richness-equalized native network. This allowed us to compare network structure between yard types independent of differences in plant diversity.

3. *Influence of dominant generalists* - In the third analysis, we examined the structural dependence of networks on dominant generalist floral visitors by removing all interactions involving *Apis mellifera*. We repeated the bootstrapping procedure described above (sampling 10 plant species per network) to generate richness-equalized native networks without *Apis mellifera* interactions. Comparing these networks to the corresponding *Apis mellifera*-removed conventional networks allowed us to assess how more specialized floral visitors contributed to network stability and complexity.

For each network, we calculated a standard suite of structural metrics to quantify interaction complexity and stability. These included connectance (the proportion of realized interactions relative to all possible interactions), modularity (the degree to which interactions form cohesive subgroups or modules), clustering coefficient (the tendency for nodes to form tightly interconnected clusters), and robustness (network resilience to sequential node loss). We also calculated degree centrality and weighted degree for each plant species and floral-visitor family to identify highly connected or influential nodes within the networks. Higher centrality reflects taxa that interact with many partners and thus play a disproportionate role in structuring plant–pollinator interactions.

Objective 5- Floral-use versus availability

To assess whether floral visitors used plant taxa more than expected based on their availability, we compared proportional floral availability (“availability”) to proportional visitation (“use”) at both the plant genus and species levels. We first summarized total floral availability and total visitation across all taxa and then calculated proportional values within each plant-origin category (native vs. non-native). For each plant genus, we quantified proportional availability as the proportion of survey events in which that genus was recorded as flowering (presence/absence of open blooms). For each floral-visitor genus, we quantified proportional use as the proportion of total floral visits attributed to that insect or bird family within the same plant-origin category. After computing proportional availability and proportional use for each taxon, we quantified their deviation as the percentage-point difference:

$$(\text{Use} - \text{Availability}) \times 100$$

The resulting values represented the magnitude of over- or under-utilization relative to expected use based on availability. To ensure robust inference, we excluded rare taxa that contributed <1% of floral records or <1% of total visits, or that had fewer than five floral records and fewer than ten visits. We then ranked genera and species by the absolute uplift in proportional use over availability (i.e., the largest positive percentage-point differences) and highlighted the 15 taxa exhibiting the strongest disproportionate use. To test whether overall floral use differed from availability, we conducted χ^2 goodness-of-fit tests at two scales in the pooled dataset: (1) across all genera and (2) between native and non-native plants.

Results

We recorded 5,109 floral-visitor interactions across 31 residential yards (18 paired sites, 13 unpaired native yards). These interactions were distributed across 108 plant genera, with visitation strongly concentrated among a subset of taxa. *Eriogonum* (buckwheat) was the most frequently visited genus, accounting for 1,368 interactions (26.8% of total), followed by *Salvia* (sage) with 429 interactions (8.4%), *Helianthus* (sunflower) with 274 interactions (5.4%), and *Encelia* (brittlebush) with 251 interactions (4.9%) (Table S1). Of all recorded floral interactions, the majority occurred on native plants (4,237 interactions, 83%) compared to nonnative plants (872 interactions, 17%). Floral visitors represented 17 families, and insect visitors dominated observations. Apidae (bees) was the most abundant group with 3,276 interactions (64.1% of total), followed by Syrphidae (hoverflies) with 1,380 interactions (27.0%), and Calliphoridae (blowflies) with 110 interactions (2.2%) (Table S2). Lepidopteran visitors were relatively uncommon, collectively accounting for approximately 3% of all interactions across multiple families (e.g., Lycaenidae, Pieridae, Hesperidae, and Nymphalidae). Native plants supported the majority of floral visitation, accounting for 4,237 interactions (82.9%), compared to 872 interactions (17.1%) on nonnative plants (Table S1). Hummingbirds (Trochilidae) accounted for 26 interactions (0.5% of total) and were observed visiting 18 plant species, including *Salvia*, *Penstemon*, *Eriogonum*, and *Encelia*.

Objective 1- Paired yard analysis

Native yards supported higher floral-visitor abundance than conventional yards (ratio=2.36 times higher than in conventional yards, $p=0.01$) and showed a trend toward higher floral-visitor richness (ratio=1.40, $p=0.09$). Plant community characteristics were greater in native yards:

total plant richness (ratio=2.85, $p<0.01$), native plant richness (ratio=19.1, $p<0.01$), total flower-color richness (ratio=2.03, $p=0.01$), and native flower-color richness (ratio=8.71, $p<0.01$), all reflecting large effect sizes (Cohen's $d>0.9$) (Fig. 2). In contrast, non-native plant richness ($p=0.20$) and non-native flower-color richness ($p=0.29$) did not differ between yard types. Among environmental covariates, temperature was 0.9 °C higher in conventional yards ($p=0.01$), whereas air quality ($p=0.22$), house price ($p=0.51$), and yard size ($p=0.68$) showed no significant differences (Fig. 2).

Floral-visitor community composition differed between native and conventional yards (PERMANOVA, $F_{1,16} = 2.63$, $R^2 = 0.14$, $p=0.04$, Fig. S1). Dispersion also differed between yard types (permutational test of multivariate dispersion, $p=0.03$), indicating that native and conventional yards varied in both composition and within-group heterogeneity (Fig. S1).

Objective 2- Relationships between environmental features and floral visitors

Floral-visitor abundance The best-supported predictor of floral-visitor abundance was total flower-color richness (quadratic model: $\beta_1 = 0.74$, $p<0.01$; $\beta_2 = -0.035$, $p<0.01$), which increased sharply at low to intermediate color diversity before declining at the highest levels (Fig. 3A). This quadratic pattern was influenced by one high-color, low-abundance observation.

and should be interpreted cautiously. On the response scale, each additional flower-color category at intermediate diversity was associated with an approximate doubling of floral-visitor abundance ($\exp(\beta_1) \approx 2.1$) before declining at higher color richness. Other models showed positive relationships but weaker support ($\Delta\text{AICc}>2$): total plant richness ($\Delta\text{AICc}=3.95$, $\beta_1 = 0.08$, $p<0.01$, Fig. 3B), yard size ($\Delta\text{AICc}=6.92$, $\beta_1 = 0.01$, $p=0.02$), native plant richness ($\Delta\text{AICc}=7.57$, $\beta_1 = 0.06$, $p=0.02$), native flower-color richness ($\Delta\text{AICc}=7.58$, $\beta_1 = 0.15$, $p=0.01$), and log(house price) ($\Delta\text{AICc}=8.38$, $\beta_1 = 1.23$, $p=0.03$, Fig. 3C). The positive association with house price suggests a luxury effect on pollinator visitation in urban systems. Temperature showed a weak positive trend ($\Delta\text{AICc}=10.22$, $\beta_1 = 0.10$, $p=0.10$). Non-native plant richness, non-native flower-color richness, and air quality were not associated with floral-visitor abundance ($\Delta\text{AICc}>9.28$).

Floral-visitor richness The best-supported predictor of floral-visitor richness was total flowering-plant richness ($\beta_1 = 0.22$, $p=0.02$, Fig. 3D). Interpreted on the response scale, each additional flowering plant species was associated with a 25% increase in floral-visitor richness ($\exp(\beta_1)=1.25$).

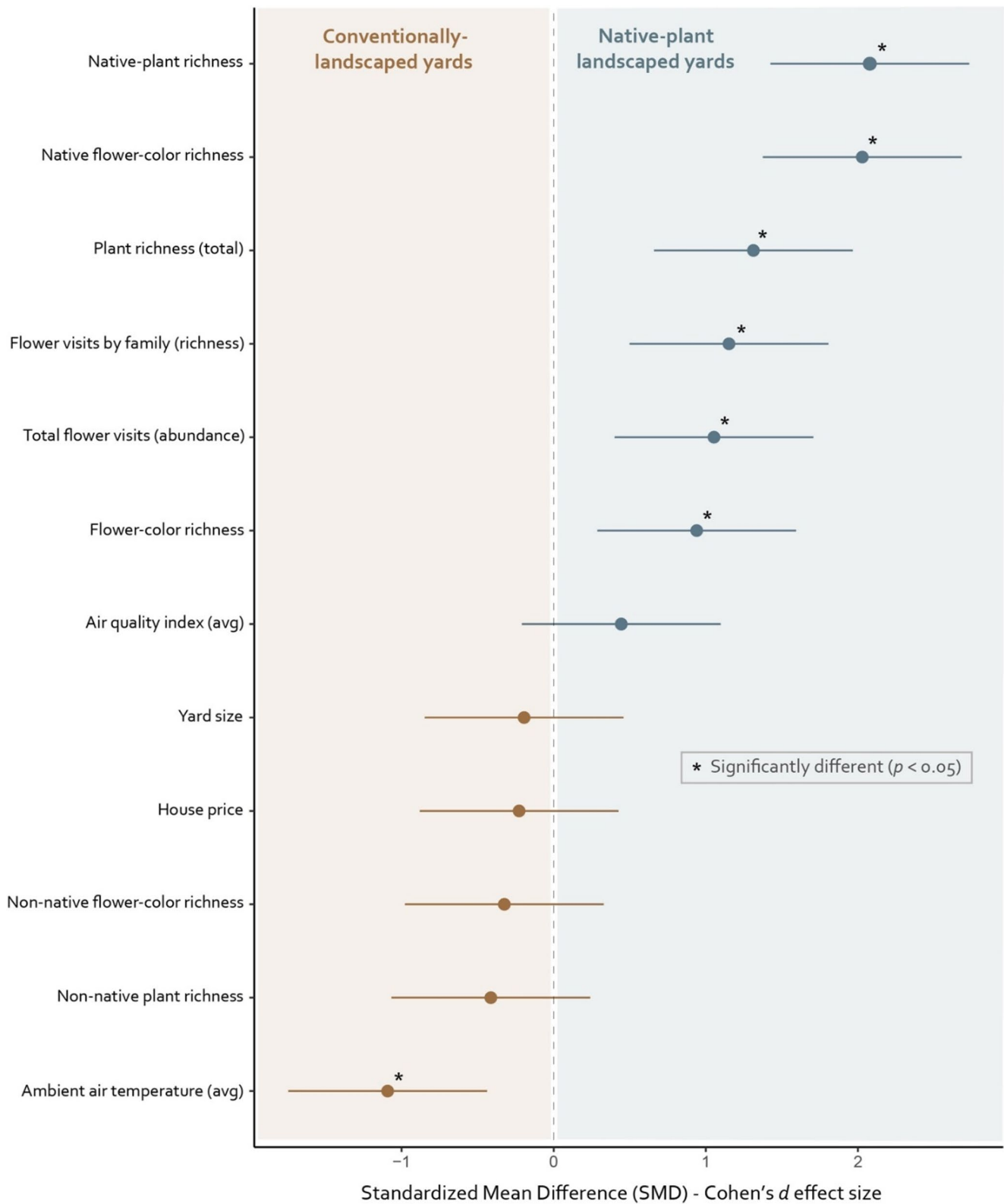
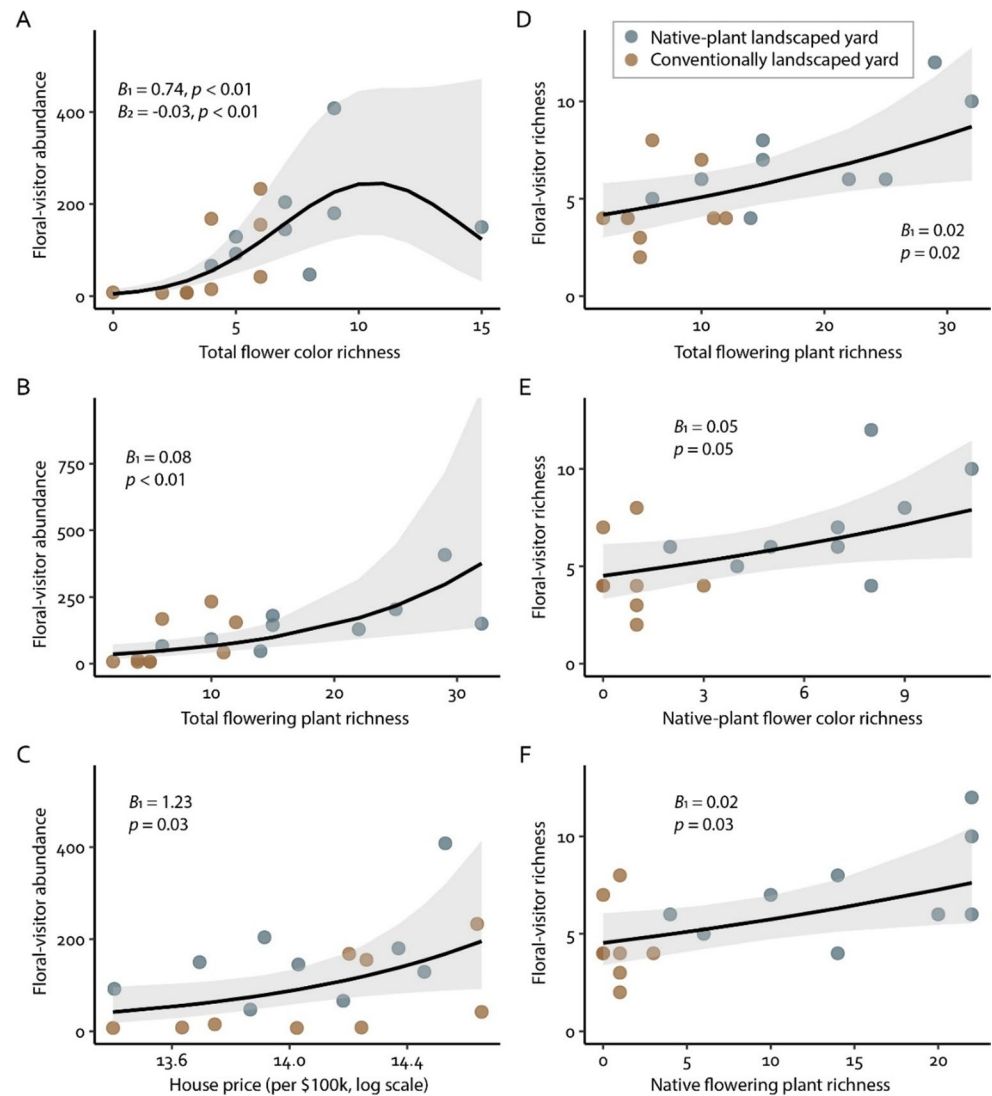


Fig. 2 Effect sizes (Cohen's *d*) comparing plant and pollinator variables between conventionally landscaped and native-landscaped yards. Dot-and-error-bar plot portrays standardized mean differences

($\pm 95\%$ CI) for response variables. Asterisks denote significant differences between yard types ($p < 0.05$). Positive values (green dots) indicate higher means in native-landscaped yards

Fig. 3 Relationships between environmental features and floral-visitor abundance (A, B, C) and richness (D, E, F) in paired native and conventional yards. We fitted abundance models using negative binomial GLMMs and richness models using Poisson GLMMs with pair ID as a random effect and survey duration as an offset. Black lines show model predictions with 95% confidence intervals (gray shading). Green points represent native-plant yards; brown points represent conventionally landscaped yards. Reported β coefficients are fixed-effect estimates on the log scale, with associated p-values. Panel A includes a quadratic term (β_2); all other panels show linear relationships



Other good fitting models explaining floral-visitor richness were native plant richness ($\Delta\text{AICc}=1.00$, $\beta_1 = 0.21$, $p=0.03$, Fig. 3E) and native flower-color richness ($\Delta\text{AICc}=1.76$, $\beta_1 = 0.19$, $p=0.05$, Fig. 3F). No other models outperformed the intercept-only model ($\Delta\text{AICc}=2.26$), and no additional predictors were significantly related to floral-visitor richness.

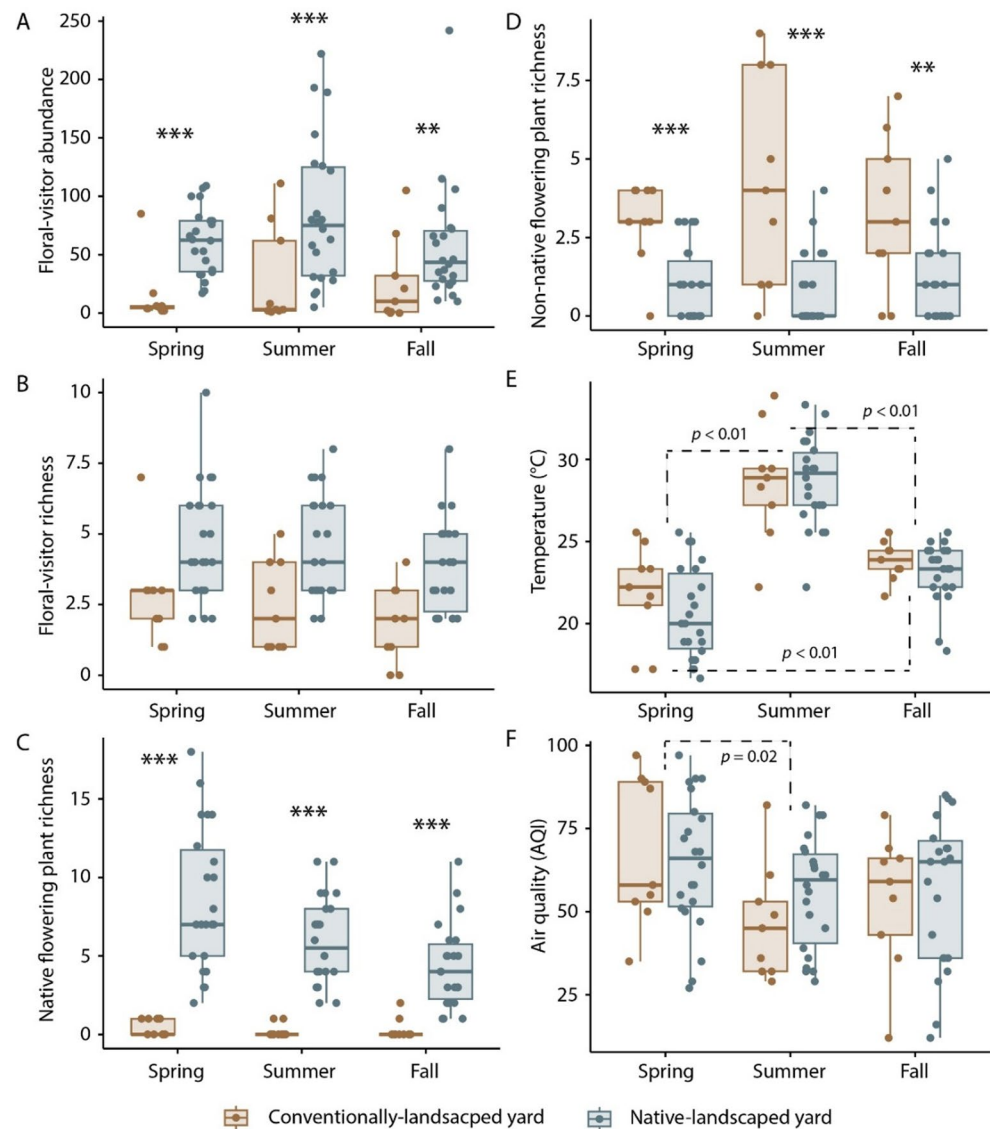
Objective 3 - Plant-floral-visitor interactions across seasons

Native yards supported significantly higher floral-visitor abundance than conventional yards across all seasons ($4.2\times$ higher averaged across visits, $p<0.01$), with no significant Type $\times$ Visit interaction ($p=0.76$), indicating this effect was consistent across spring, summer, and fall (Fig. 4A). Pair-wise comparisons showed native yards had $4.4\times$ higher abundance in spring ($p<0.01$), $4.8\times$ higher in summer ($p<0.01$),

and $3.6\times$ higher in fall ($p<0.01$, Fig. S2). Abundance did not vary significantly across visits ($p=0.55$), though marginal means showed a numeric peak in summer (33.0 visits/min) compared to spring ($24.3/\text{min}$) and fall ($32.9/\text{min}$, Fig. S2). Floral-visitor richness showed weaker yard-type effects (Fig. 4B). Native yards averaged $1.5\times$ higher richness than conventional yards ($p<0.01$), but this effect varied seasonally (Type $\times$ Visit $p=0.30$, Fig. S2). Richness did not differ between yard types in spring ($p=0.71$), but native yards supported higher richness in summer ($1.6\times$ higher, $p=0.05$) and fall ($1.8\times$ higher, $p=0.03$). Richness did not vary significantly across visits ($p=0.67$, Fig. S2).

Native yards supported $1.7\times$ more flowering plant species across seasons than conventional yards ($p=0.04$), driven almost entirely by native flowering plants, which were $14.5\times$ more abundant in native yards ($p<0.01$, Fig. 4C). In contrast, conventional yards had $5.8\times$ more non-native flowering plants ($p<0.01$, Fig. 4D). Flowering plant richness did

Fig. 4 Temporal patterns in floral-visitor activity, floral resources, and environmental conditions across three seasonal visits (Spring: April–May, Summer: June–July, Fall: August–September) in native-plant (green) and conventionally landscaped yards in Los Angeles, CA. Panels show: (A) floral-visitor abundance, (B) floral-visitor richness, (C) number of native flowering plant species, (D) number of non-native flowering plant species, (E) temperature (°C), and (F) air quality (AQI). Boxplots show median values (horizontal bars), interquartile ranges (boxes), and distributions (whiskers and points). Asterisks indicate significant differences between yard types within each visit: ** $p < 0.01$, *** $p < 0.001$ (Type $\times$ Visit GLMMs). Dashed lines with p-values in E and F denote significant seasonal changes (main effect of Visit)



not vary significantly across visits for any plant category (all $p > 0.66$). Environmental conditions varied seasonally but not by yard type (Fig. S2). Temperature peaked in summer (28.6 °C) compared to spring (21.9 °C) and fall (23.1 °C) ($p < 0.01$, Fig. S2), with no difference between yard types ($p = 0.28$) (Fig. 4E). Air quality (AQI) was significantly worse in spring (66.2) than summer (50.4, $p = 0.02$) likely due to seasonal spring marine layer inversions typical of coastal Southern California but did not differ between yard types ($p = 0.57$) (Fig. 4F, S2).

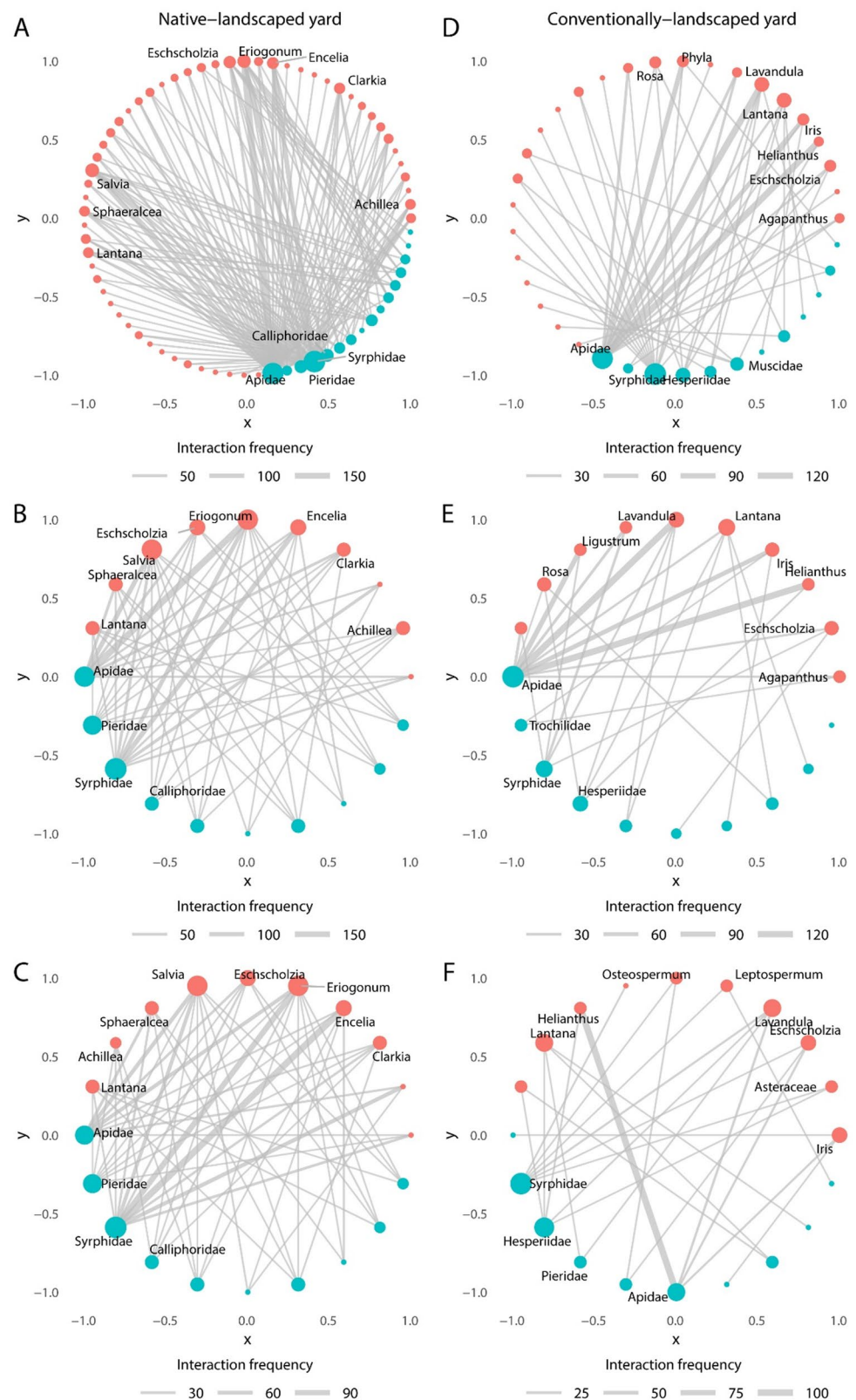
Objective 4- Floral-visitor networks

Paired-yard network comparison (Full networks) The pooled native-yard interaction network contained more realized plant–visitor interactions than the conventional-yard network (140 vs. 48 edges) and exhibited higher overall

connectivity [mean degree (average interactions per node) 3.94 vs. 2.59 (Fig. 5A, D). This indicates that native yards supported richer and more complex interaction webs, with individual plants and insect families participating in more partners and linking more strongly across the network. In contrast, modularity was higher in the conventional network ($Q = 0.48$ vs. 0.35), suggesting greater compartmentalization of interactions into discrete clusters (Fig. 5A, D) where subsets of plants and floral visitors interact primarily among themselves rather than across the broader network.

Plant-origin effects (Top 10 species networks) After equalizing plant-species richness through bootstrapping, native yards still produced interaction networks that were markedly more connected than their conventional counterparts (Fig. 5B, E). Among the 10 most frequently visited plant species, native networks contained twice as many edges (56

Fig. 5 Bipartite interaction networks comparing native and conventional residential yards, with plant species (red circles) and floral-visitor families (blue circles) arranged in circular layouts. Circle size is proportional to the total number of interactions per node, and edge thickness represents interaction frequency. Panels **A** and **D** show full networks including all observed interactions for native and conventional yards, respectively. Panels **B** and **E** show reduced networks including the top 10 most frequently interacting plant species for native (B) and conventional (E) yards. Panels **C** and **F** show equivalent reduced networks following removal of *Apis mellifera* interactions for native (C) and conventional (F) yards. Together, these networks illustrate differences in interaction structure, connectivity, and reliance on dominant generalist species between yard types and highlight the greater connectivity and resilience of native-yard networks relative to conventional yards



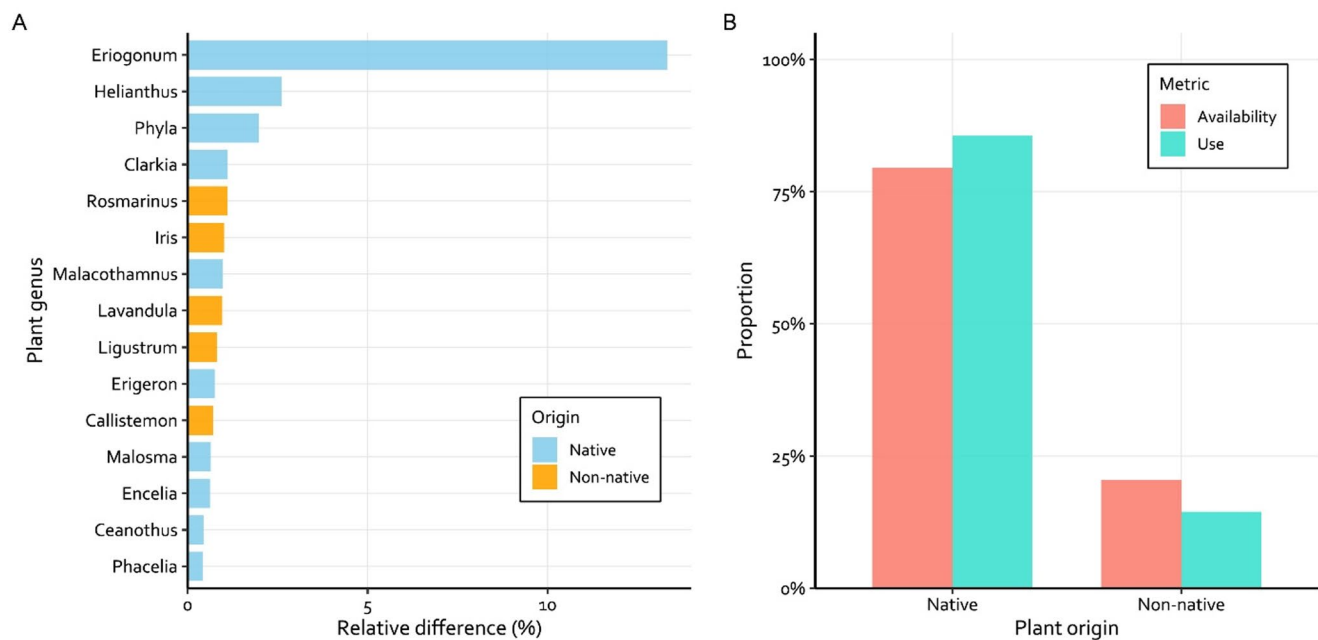


Fig. 6 (A) Top 15 plant genera used disproportionately more than their availability in residential yards, ranked by relative difference between proportional use and proportional availability. Bars are colored by plant origin: native (blue) and non-native (orange). (B) Comparison of proportional availability versus use of native and non-native plants

vs. 28), and mean degree was correspondingly higher (5.6 vs. 2.8), indicating that dominant native plants supported roughly twice as many floral-visitor interactions as dominant conventional plants (Fig. 5B, E). Modularity remained low in the native network ($Q=0.17$) but was notably higher in the conventional network ($Q=0.36$), reinforcing a pattern of tightly interwoven interactions in native yards and more compartmentalized, compositionally narrow interactions in conventional yards (Fig. 5B, E).

Influence of dominant generalists (*Apis mellifera*-removed networks) Removing all *Apis mellifera* interactions revealed strong divergence in underlying network structure between yard types (Fig. 5C, F). In the absence of this dominant bee species, native networks retained higher connectivity (55 edges; mean degree 5.5), indicating that native plants attracted a diverse array of non-*Apis mellifera* floral visitors that continued to maintain network cohesion (Fig. 5C, F). In contrast, conventional networks collapsed to a sparse and highly modular structure (25 edges; mean degree 2.5; modularity $Q=0.44$), suggesting that *Apis mellifera* interactions were disproportionately contributing to network structure in conventional yards (Fig. 5C, F). Once removed, the limited redundancy and narrower visitor pools associated with ornamental plants became apparent. Native networks, by contrast, remained relatively cohesive and low in modularity ($Q=0.16$), demonstrating that native

across all yards. Native plants were used more than their availability (88% of floral-visitor interactions vs. 80% availability), while non-native plants were used less than their availability (12% of interactions vs. 20% availability), suggesting selective feeding on native plants by floral visitors

landscaping enhanced network stability even in the absence of *Apis mellifera* (Fig. 5C, F).

Objective 5- Floral-use versus availability Floral visitors exhibited strong, non-random use of plant taxa, with clear mismatches between floral availability and visitation patterns (χ^2 tests across genera and plant-origin categories, both $p<0.01$). At the genus scale, several native taxa showed exceptionally strong over-utilization relative to availability (Fig. 6A, Table S3). Most notably *Eriogonum* received ~13% more visitation than expected, followed by *Helianthus*, *Phyla*, and *Clarkia* (Fig. 6A, Table S3). In contrast, most non-native genera were either used in proportion to their availability or were modestly under-utilized (Fig. 6A, Table S3). Further, across the full dataset, native plants composed roughly 80% of available flowering resources, yet received nearly 90% of all floral visits, whereas non-native plants accounted for ~20% of availability but only ~10% of total visits (Fig. 6B). This pattern demonstrated a significant overall preference for native plants at the community level.

Discussion

Cities present unique opportunities for conservation through intentional landscape design and community engagement. Because residential yards comprise a substantial portion of

urban greenspace, they can function as ecologically valuable habitat when managed with biodiversity in mind (Goddard et al. 2010b; Aronson et al. 2017; Baldock 2020). Our results show that native-plant landscaping can translate these opportunities into measurable ecological benefits, supporting higher floral richness and greater flower-color diversity, which in turn attract more diverse and abundant insect and bird floral visitor communities. Our results build on previous investigations on the importance of native plants to urban pollinators (e.g., Burghardt et al. 2009; Frankie et al. 2009; Tartaglia and Aronson 2024; Lilkendey et al. 2025). These effects were not fleeting as native yards maintained stronger seasonal resilience, with bird and insect floral visitation remaining relatively high even in late summer when conventional yards experienced marked declines (Wenzel et al. 2020). Critically, native yards harbored more diverse and stable plant-floral visitor interaction networks, characterized by higher connectivity, lower modularity, and more evenly distributed centrality among species. Together, these patterns indicate more integrated interaction networks, where species are linked through multiple partners rather than partitioned into isolated subgroups, enhancing redundancy and the capacity of the system to maintain function under disturbance. This suggests that native landscaping promotes not just species counts, but functional ecological complexity (Albrecht et al. 2012; Kaiser-Bunbury et al. 2017). Taken together, our findings reinforce the idea that local stewardship can scale up to regional conservation impacts (Dearborn and Kark 2010; Goddard et al. 2010a; Aronson et al. 2017; Rega-Brotsky et al. 2022). In an era of accelerating urban growth and climate disruption, empowering homeowners and communities to create ecologically functional yards offers a hopeful and actionable path toward more biodiverse and resilient cities (Goddard et al. 2010b; Aronson et al. 2016; Bennett and Lovell 2019; Mata et al. 2021).

At the yard scale, we found consistent differences between native and conventionally landscaped yards, with native yards supporting greater biodiversity across both plant and floral visitor communities. These differences were broad-based. Native yards had higher total and native plant richness, more diverse floral color palettes, and greater floral-visitor richness and abundance for insects, as well as higher hummingbird visitation. These findings align with previous studies showing that structurally and botanically complex yards can support more diverse wildlife (Goddard et al. 2010b; Lerman and Warren 2011), and they extend recent work in Southern California suggesting that even modestly sized native plantings can enhance habitat use by birds in residential settings (Smallwood and Wood 2023). Unlike many landscape-scale studies, our paired design controlled for variation in income levels, yard size, and regional context, allowing for a more direct comparison

of yard-level vegetation choices. Interestingly, while conventional yards were not necessarily depauperate in plant species or flower color overall, their composition did not translate into higher biodiversity for insect and bird flower visitors. This pattern is consistent with recent work in Los Angeles showing that cultivated urban plant communities are often highly diverse but dominated by non-native species selected through horticultural availability and resident preferences, resulting in shifts in community composition and functional traits relative to remnant ecosystems (Avolio et al. 2020). This contrast supports prior findings that native flora may play a disproportionately important role in supporting urban ecological communities (e.g., Burghardt et al. 2009; Narango 2018), and reinforces the value of local, yard-scale interventions (Goddard et al. 2010b).

Understanding the ecological drivers that shape biodiversity in urban environments is essential for designing more effective conservation interventions. Our results suggested that plant richness, particularly the total number of flowering species, was the single most important predictor of both floral-visitor abundance and richness, outperforming all other environmental and housing variables. This aligns with current knowledge that floral resource diversity is a primary determinant of pollinator community structure in cities (Mata et al. 2015; Baldock et al. 2015; Hall et al. 2017; Yong 2024; Lilkendey et al. 2025). The strong performance of both native and total flower color richness as predictors further underscores the importance of visual floral diversity, which likely facilitates foraging across diverse pollinator functional groups, including birds, bees, and flies (Fukase and Simons 2016; Ballantyne et al. 2017; Lilkendey et al. 2025). Yard size was also a significant univariate predictor, reinforcing that the species–area relationship remains a relevant pattern in urban ecological systems, particularly when examining plant–pollinator interactions (Lepczyk et al. 2017; Aronson et al. 2017; Mata et al. 2021; Lilkendey et al. 2025).

These results extend previous work on the “luxury effect” (Hope et al. 2003; Leong et al. 2018). While this pattern has traditionally been documented in terms of vegetation cover or species presence, our study extends it to functional ecological interactions, showing that yards in higher-income areas tend to support a greater abundance of insect and bird floral visitors. However, by isolating native plant richness and floral diversity as the key ecological drivers, we provide compelling evidence that this disparity is not inevitable. In fact, several of the most florally diverse and interaction-rich yards in our study were located in lower-income neighborhoods in central Los Angeles. We note that we did not explicitly quantify proximity to large urban greenspaces or other potential source habitats (see Prudic et al. 2022), which may also influence floral visitor communities through

broader landscape or metapopulation dynamics. Similar spatial effects have been documented for birds in this system (Smallwood and Wood 2023). Nevertheless, our findings indicate that the ecological outcomes often associated with the luxury effect can likely be achieved through targeted interventions, specifically the replacement of conventional landscaping with native flowering plants.

Our results support prior work demonstrating that lower-income communities in Los Angeles often suffer from reduced green cover and limited access to biodiversity (Wood and Esaian 2020; Wood et al. 2024). Nevertheless, our findings offer a hopeful message that equitable, small-scale interventions can generate substantial ecological value, even in highly urbanized and economically constrained settings. This is especially important given ongoing concerns about food insecurity and the increasing role that urban community gardens, like those championed by community stewards such as Ron Finley, play in neighborhood resilience (Frangos and Imbesi 2014). While our study did not directly evaluate crop pollination, attracting diverse floral visitors into neighborhoods via native landscaping may provide indirect ecological benefits to food plants grown in home or community gardens. Indeed, recent work has shown that integrating native flowering species into urban garden plots enhances pollinator visitation and potential pollination services in cities (Pawelek et al. 2009). In this way, native landscaping may contribute not only to biodiversity goals, but to community food systems and environmental justice.

Seasonal shifts in plant–pollinator interactions are a defining feature of temperate ecosystems (Geslin et al. 2013), and our findings show that even in small urban yards, these dynamics are both detectable and ecologically meaningful. Native yards supported significantly higher floral-visitor abundance than conventional yards consistently across spring, summer, and fall, suggesting that native plants provide reliable, season-long resources rather than ephemeral floral pulses (Wenzel et al. 2020). This temporal stability may be particularly important for maintaining pollinator populations through periods of resource scarcity, when conventional landscapes offer limited foraging opportunities (Baldock et al. 2015; Sánchez and Lara 2024).

The phenological mismatch between native and non-native flowering plants revealed an important ecological trade-off. Native flowering plant richness was substantially higher in native yards during spring, aligning with the Mediterranean-climate strategy of early-season reproduction before summer drought (Shmida and Dafni 1989). In contrast, conventional yards maintained higher non-native flowering plant richness that peaked in summer and fall – likely sustained by irrigation – yet this extended bloom period did not translate into increased floral visitation. This suggests that pollinators in urban Southern California may

be adapted to native plant phenologies and floral traits, consistent with observations that co-evolved plant–pollinator relationships exhibit higher fidelity and effectiveness than novel interactions (Ollerton et al. 2011; Geslin et al. 2013). The inability of non-native ornamental blooms to compensate for absent native resources underscores a fundamental principle: ecological function cannot be simply replaced by visual abundance (Burghardt et al. 2009; Frankie et al. 2009; Mata et al. 2021).

Our network analyses revealed that native landscaping restores not only floral-visitor abundance but also the structural complexity underpinning ecological resilience. Native yards supported networks with higher connectivity and lower modularity, which are signatures of interaction redundancy that buffered communities against species loss and environmental perturbation (Bascompte 2007; Tylianakis et al. 2010). In contrast, conventional yards exhibited simplified, highly modular networks dominated by generalist pollinators, particularly honeybees. This structural resilience may be particularly important in Southern California, where climate instability, including prolonged drought and intensifying heat extremes, threatens urban ecological communities (Diffenbaugh et al. 2015; Mann and Gleick 2015; Williams et al. 2020). Networks with high redundancy can maintain function even when individual species are lost or suppressed by environmental stress (Memmott et al. 2004; Tylianakis et al. 2010), because multiple floral visitors can share the same plant resources, and many plants may be visited by more than one species. This overlap reduces reliance on any single interaction. Conventional yards, by contrast, appear vulnerable to cascading collapse if dominant generalists, such as *Apis mellifera* decline, a realistic scenario given documented honeybee sensitivity to anthropogenic stressors (Goulson et al. 2015).

The use-versus-availability analyses revealed important native plants, including species such as California buckwheat (*Eriogonum fasciculatum*) that disproportionately anchored resilient networks. This underscores the value of moving beyond species counts toward functional metrics like interaction richness and network redundancy as indicators of urban restoration success (Kaiser-Bunbury et al. 2017). Future work linking network structure to pollination outcomes (seed set, reproductive success) will be critical for demonstrating how yard-scale interventions translate into population-level ecological benefits.

Conclusion and Applications

Our study provided support that native-plant landscaping in residential yards significantly enhanced urban pollinator communities across multiple dimensions: floral-visitor

abundance and richness, seasonal stability, and network complexity. Based on our findings, we recommend the following applications for urban planning and conservation:

1. *Prioritize native plant conversions* – Native yards supported 4× higher floral-visitor abundance and more complex interaction networks than conventional yards. Importantly, scaling our observed network differences suggests that even modest adoption could yield meaningful ecological gains. For example, converting just 1% of residential yards to native landscaping would add 100 additional plant–floral-visitor interactions per 1,000 yards, based on observed differences between yard types (~10 additional interactions per converted yard). Scaled across a large city such as Los Angeles, this represents tens of thousands of additional ecological interactions distributed across the urban landscape (Table S4). Cities should incentivize native conversions through rebates and landscape design programs, with priority given to low-income neighborhoods where baseline native cover is lowest and ecological return on investment is highest.
2. *Ensure equitable access to native plant conversions* – Native plant benefits persisted across the housing-price gradient, with comparable increases in floral-visitor richness regardless of neighborhood income. This challenges the assumption that biodiversity gains require affluent communities or high-resource investments. While structural constraints such as limited planting space and maintenance demands can complicate large-scale greening efforts in low-income neighborhoods, small-scale native plantings (e.g., shrubs, grasses, forbs) are highly feasible and ecologically meaningful. Cities should prioritize native landscaping incentives in low-income areas that historically lack green amenities, demonstrating that accessible, affordable interventions can achieve substantial ecological outcomes.
3. *Feature functionally important native species* – Plants like California buckwheat and white sage disproportionately anchored pollinator networks in our study. Regional plant palettes should emphasize high-impact native species identified through local monitoring (Table S3).
4. *Avoid over-reliance on non-native ornamentals* – Non-native flowering plants contributed minimally to pollinator visitation and were associated with simplified network structure. Total floral cover or richness alone is not an adequate proxy for ecological value.
5. *Support seasonal resource continuity* – Native plants provided consistent pollinator resources across spring, summer, and fall. Planting designs should prioritize staggered bloom periods of native species rather than

extended flowering of ornamentals. Incorporating high native plant richness and a diversity of floral traits, including variation in color, may further support diverse and abundant pollinator communities.

6. *Integrate functional metrics into planning* – Moving beyond species counts, cities should track interaction richness and network redundancy as indicators of ecological function in biodiversity monitoring and green infrastructure projects.
7. *Plan for climate resilience* – Native yards exhibited network features (redundancy, low modularity) associated with ecological resilience to environmental stress. These should be integrated into urban climate adaptation strategies.
8. *Leverage scalable monitoring* – Our yard-based methods require minimal training and could be implemented through community science programs, providing cities with fine-scale data to guide adaptive management.

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Author contributions E. R, N.S, and E.M.W conceived the research and designed the study; E.R conducted fieldwork and data development; E.R. and E.M.W analyzed data and produced figures; E.R, N.S, and E.M.W contributed to writing and editing the manuscript.

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Data availability If the paper is accepted for publication, data will be permanently archived in the Dryad repository.

Declarations

Ethics statement No animals were handled during this research. Further, no human subjects were included in this work. This research was conducted in compliance with the Guidelines to the Use of Wild Birds in Research. This project has no conflicts of interest.

Open research statement If the paper is accepted for publication, data will be permanently archived in the Dryad repository.

Competing interests The authors declare no competing interests.

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