Native bee genus diversity within bee-friendly urban gardens varies little along an urbanization gradient

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Abstract

Urban bee populations are threatened by ongoing habitat loss and fragmentation. Bee-friendly gardens with abundant forage and nesting resources may help offset these pressures, but attributes of the broader urban landscape could also play an important role. We explored bee diversity within 32 bee-friendly gardens distributed throughout the city of Kelowna, British Columbia. Our objectives were to (i) estimate bee genus abundance, richness, and composition within the gardens, and (ii) investigate how these facets of diversity varied with plant diversity, garden size, and degree of urbanization within 300m of the garden. In total, over 10 sampling days per garden, we observed 19 genera—about half of those known from the region—primarily within the Halictidae family (66.4% of total abundance). Neither bee genus richness nor total bee abundance was associated with any of the explanatory variables, whereas a minor (though statistically significant) percentage of the among-site variance in genus composition was uniquely accounted for by the urbanization gradient (6.8% with rare genera excluded, 5.8% when included). Our study is the first to evaluate urban bee diversity within the Okanagan diversity hotspot and our findings add to previous studies elsewhere demonstrating inconsistent effects of garden and landscape-scale characteristics on bee diversity.

Keywords: bee diversity, urban gardens, urbanization, landscape effects, hotspot

Introduction

Bee populations can be negatively impacted by urbanization activities, including habitat fragmentation, reduced food and nesting site availability, and increased exposure to anthropogenic stressors (Hernandez et al., 2009; Cardoso and Gonçalves, 2018; Pereira et al., 2021; Chau et al., 2023; Pfeiffer et al., 2023). Conversely, fragmented green spaces within cities may act as refuges for bees (Banaszak-Cibicka et al., 2016; Mach and Potter, 2018; Conflitti et al., 2022), supplying necessary resources within an otherwise unsuitable environment (Tommasi et al., 2004). Urban gardens are especially important, with floral diversity being a key element in attracting and maintaining diverse bee communities (Hall et al., 2017; Birdshire et al., 2020).

Situated in the Western Interior Basin of southern British Columbia, Canada, the Okanagan Valley hosts a great diversity of native bees (Sheffield and Heron, 2018), including approximately half of Canada's known bee species, one-third of which are thought to be unique to this region, nationally (Heron and Sheffield, 2015). There were 411 confirmed species in the region as of 2018 (Sheffield and Heron, 2018), spanning at least 38 native genera and 5 families. While the Okanagan hosts some of Canada's greatest bee diversity, it may also house its most vulnerable species, as much of the valley's low-elevation grassland habitats have been lost to urban and agricultural development. Thus, immense conservation potential currently exists and conservation efforts are gaining momentum in the region (Heron and Sheffield, 2015).

A series of initiatives have been launched globally within cities, aimed at understanding pollinator services, addressing conservation needs, and improving or restoring networks of urban pollinator habitat (e.g., Bee Cities, Bee Campuses, https://beecitycanada.org/). The Border Free Bees initiative is a key example within the Okanagan. It kick-started the Bee Ambassador Project in 2018, involving a large effort to improve pollinator habitat connectivity throughout the city of

Kelowna, one of Canada's fastest growing urban centres. To become a bee ambassador, participants must plant and maintain a garden containing pollinator-friendly flowers that are drought-tolerant, pesticide-free and encompass a blooming range from spring to fall, and must also leave bare spots in their garden to supply habitat for wild bees (Border Free Bees, 2017). More than 350 properties across the city were participating as of 2019, the vast majority being private home gardens (Border Free Bees, 2018). The potential for private gardens to contribute to urban pollinator conservation has become increasingly recognized (Levé et al., 2019; Majewska and Altizer, 2020; Persson et al., 2023), and while initiatives such as Border Free Bees have informed residents on creating their own "pollinator friendly" gardens, the effectiveness of this strategy is yet to be fully understood (Fukase and Simons, 2016). As the Okanagan presents a unique landscape within Canada and the Bee Ambassador Project has garnered substantial support, we sought to quantify bee diversity within a sample of putatively bee-friendly urban gardens.

Both local and landscape-scale factors can influence the effectiveness of a garden in providing urban pollinators with adequate resources. Local features are associated with the characteristics of a garden itself, such as floral diversity and garden size (Quistberg et al., 2016). Landscape features are typically associated with the location of a garden and surrounding land cover types. While the presence of green spaces in proximity to a habitat can promote bee species richness (Pardee and Philpott, 2014), the increased presence of impervious surface cover generally reduces bee richness and abundance (Quistberg et al., 2016). These impacts may be mitigated with increased local floral resources (Baldock, 2020), which can be better accommodated with increased garden area. In examining the local and landscape characteristics of urban gardens, it is also important to consider how they may affect the diversity of taxa, including taxonomic composition, within and among gardens. While many studies have found that increased urbanization tends to be associated with lower richness (Threlfall et al., 2015; Simao et al., 2018), others have concluded that richness may actually increase alongside urbanization (Martins et al., 2017) or that there is minimal change across an urban gradient (Fowler, 2014). Additionally, different urban green areas including gardens have different influences on pollinator species composition (Fischer et al., 2016; Dylewski et al., 2019). Thus, it remains unclear how urbanization influences bee taxonomic richness and composition, particularly in conjunction with local-scale factors. Previous work has found that assemblages of bee communities typically respond more positively to local-scale characteristics such as floral richness, with land-cover composition relative to increased urbanization often negatively impacting species within a community (Martins et al., 2017; Landsman et al., 2019).

Our general objective was to quantify the diversity of native bees visiting urban "Bee Ambassador" gardens within Kelowna, BC. We addressed two specific objectives; to explore how the (i) richness and (ii) taxonomic composition of native bees, both quantified at the genus level, varied in relation to plant diversity, garden size, and urbanization measured at a broader spatial scale. We were specifically interested in whether bee diversity varied systematically with degree of urbanization, despite what we assumed at the outset to be generally favourable local conditions across the sampled bee-friendly gardens.

Methods

Site selection and degree of urbanization

Of the >350 Bee Ambassador sites across Kelowna and its vicinity, we surveyed 32 gardens corresponding to those Bee Ambassadors who responded positively to requests to survey their garden (Fig. 1). We acknowledge that being restricted to a sample size of 32 may have limited our statistical power.

We conducted geospatial analyses using R version 4.5.1 “Great Square Root” (R Core Team, 2025) to calculate distances between gardens, and to quantify an indicator of urbanization that was used as a predictor variable in subsequent analyses. Fully annotated code is available at the public archive on OSF (https://doi.org/10.17605/OSF.IO/6YNCV; Pither et al., 2025). We used the following R packages within these geospatial analyses: *terra* (version 1.8-60; Hijmans, 2025), *osmdata* (version 0.3.0.9004; Padgham et al. 2017), *sf* (version 1.0-21; Pebesma and Bivand, 2023), *rnaturalearth* (version 1.1.0; Massicotte and South, 2025), *rnaturalearthdata* (version 1.0.0; South et al. 2024a), *rnaturalearthhires* (version 1.0.0.9000; South et al. 2024b), *ggplot2* (version 3.5.2; Wickham, 2016), *ggspatial* (version 1.1.10; Dunnington, 2023), and *tmap* (version 4.2; Tennekes, 2018).

Using geospatial data from the City of Kelowna (Address Points, 2016-2019) and the Regional District of Central Okanagan open data (RDCO - Addresses, 2014-2020; RDCO Buildings, 2019-2020; RDCO Roads, 2014-2020), we quantified the total building area (Fig. S1) and length of road or lanes (Fig. S2) within a 300 m radius (henceforth “buffer distance”) of each ambassador garden to enable the creation of the variable urbanization index. We initially opted for a 300 m buffer distance because using larger distances would have increased non-independence (pseudoreplication) among samples owing to overlap of geospatial data. Nevertheless, as a form of sensitivity analysis (described below), we repeated all our main statistical analyses using the urbanization index derived from distances of 100m, 200m, 300m, 500m, and 700m. For our main 300 m analyses we retained all 32 sites, and note that 2 sites could be considered pseudoreplicates due to their proximity being closer than 300 m.

We performed a principal components analysis (PCA) (*princomp* function in base R) on the length of roads and building area within the given buffer distance, and extracted the first PCA component as the urbanization index. For the 300 m buffer distance this first component accounted for 80.7% of the variance.

Vegetation surveys

Local garden characteristics were measured during vegetation surveys conducted once at each site, on dates spanning July 2 to September 28, 2019. Although plant species composition was constant at all surveyed gardens (i.e. no new plantings occurred during the study), we acknowledge that the broad timespan of our vegetation sampling combined with only a single survey of each garden may have impacted our assessments of plant diversity and diminished our power to detect its potential influences on bee assemblages.

The spatial extent of the vegetation surveys depended on the size of each study garden and its proximity to other areas of garden at a site. In some cases, gardens spanning an entire yard were accounted for, while in other cases there may have only been one small garden on a property. To help account for this variation, we measured garden area at each site, accomplished in-field or using Google Earth. Study gardens ranged in size from 3.31 to 192 m^2 (Fig. S3).

We collected photographs of gardens and the plants within them to aid in plant species identification, supplemented by in-field identification efforts where possible. To obtain plant species abundance, we estimated the percentage of the total vegetation cover of a garden accounted for by each species.

Using the plant species data, we calculated plant diversity as the Hill number of order 1: ${}^{1}D_{plants} = exp\left(-\sum_{i=1}^{S} p_i \log(p_i)\right)$, where S is the number of plant species in the garden, and p_i is the proportional abundance of plant species *i*. This is equivalent to the exponential of Shannon entropy (Chao et al., 2014). We calculated this using the *diversity* function in the *vegan* package (version 2.7-1; Oksanen et al., 2025), with the argument index = "shannon", and taking the exponential of the value. We used ${}^1D_{plants}$ as one of the predictors of bee genus richness and abundance (see below).

Bee sampling and identification

Permits to collect bees were not required for this study. To obtain in-field estimates of the alpha diversity (genus richness) and abundance of bees at all study sites, we deployed a series of pan traps following the USGS Native Bee Inventory and Monitoring Lab protocol (Droege et al., 2016). The pan traps were 12.7 cm in width and 4.3 cm in depth, and one of three fluorescent colours: yellow, blue and white. These colours have been shown to be particularly effective at attracting many bee species (Grundel et al., 2011). We arranged traps in sets of three, henceforth referred to as triplets. One triplet was deployed at each site during each sampling period. The triplets consisted of one trap of each colour (Pardee and Philpott, 2014; Quistberg et al., 2016), filled ¾ with a mixture of water and Dawn Original Scented Dish Soap. They were set at a height slightly above the average height of vegetation (Normandin et al., 2017), in areas with adequate sunlight (Bates et al., 2011). To accomplish this, we attached traps to a wooden stake using three loops of galvanized strapping, which could be set at different heights using pre-drilled holes. The main sampling season spanned from June 10 through August 29, 2019, with bee sampling conducted five times throughout this period. Sampling only took place on clear or mostly clear days with little wind and a temperature of at least 15 °C. Bee taxa that are typically active early in the flight season (April/May) were likely under-sampled. Sites were grouped for surveying by proximity for logistical simplicity. The order that sites were sampled within the survey groups was randomized, following a similar approach to Simao et al. (2018). While sampling from sites randomly throughout the day (i.e., Martins et al., 2017) or in order along the urban gradient (i.e., Bates et al., 2011) would be an ideal approach in limiting temporal sampling bias, due to temporal and logistical limitations, travelling between gardens in that manner would not be an efficient method. An approach that we took to minimize temporal sampling bias was to alternate site visits between the morning and afternoon to allow for the coverage of whole foraging bee populations (Fortel et al., 2014). This was accomplished by modifying the route of travel for each sampling period.

For a given sampling foray we left traps out at each site for a period of 48 hours, after which all materials were collected. Specimens were retrieved by pouring trap contents into a sieve. We collected traps from sites in the same order in which they were placed, at approximately the same time of day. The setup and collection schedule noted above was repeated every two to three weeks throughout the sampling period, the goal being that sampling each site within this time frame would allow for the observation and collection of bees across various blooming periods throughout the season. Triplets at five gardens required relocating after initial trapping attempts due to insecure stake positioning. At these sites, only the data from subsequent visits using the repositioned traps were used. Traps were deployed at each garden for a total of 48 x 5 = 240 hours, or 10 days.

Following retrieval, we transferred trap contents into a Whirl-Pak® Standard Sterilized Bag and stored in a cooler for transport back to the lab. During the first collection period, specimens from all bowl colours within a triplet were mistakenly pooled together. For all subsequent collection periods specimens from each triplet were separated based on bowl colour. Upon returning to the lab, we filled Whirl-Paks® full of collected insects with 70% ethanol

(Droege et al., 2016) and refrigerated the Whirl-Paks® until they were ready to be processed.

In-lab processing and identification of bees began in August 2019, continuing until January 2020. We separated bee and non-bee specimens, with collected bees washed and dried following USGS protocols (Droege et al., 2016). Non-bees were not evaluated in this study. We recorded bee specimens to at least the genus level, the following data recorded for each specimen: site, collection date, family, subfamily or tribe (if applicable), genus, species (for a few select specimens, e.g. *Agapostemon virescens* (Fabricius, 1775), *Bombus navadensis* Cresson, 1874)) and bowl colour (though the latter was not considered further). Identification was largely aided by tools and techniques described in *The Bees in Your Backyard: A guide to North America's Bee* (Wilson & Messinger Carril, 2016), with confirmation from various online resources. Much of the bumble bee identification was aided by informational resources created by Sheffield and Heron (2018). The complete list of sampled individual bees and associated taxonomic information is included in the publicly available data archive. Voucher specimens were retained, pinned and stored in the lab (Biodiversity & Landscape Ecology Research Facility at the Okanagan campus of UBC), following USGS protocols (Droege et al., 2016).

As the focus of the study was on native bees, we excluded all observations of *Apis mellifera*, the European honey bee, from the main analyses. A total of 131 occurrences of *Apis mellifera* were recorded, accounting for approximately 6.6% of the total number of bees collected (native and non-native).

Statistical analyses

We conducted all statistical analyses using R version 4.5.1 "Great Square Root" (R Core Team, 2025). The data and scripts, including a fully annotated R markdown script, are openly available at OSF (https://doi.org/10.17605/OSF.IO/6YNCV; Pither et al., 2025). In addition to those specified below, we used the following packages (versions shown in scripts) to support our analyses and to maximize computational reproducibility: *magrittr* (Bache and Wickham, 2022), *janitor* (Firke, 2024), *here* (Müller, 2020), and *ggrepel* (Slovikowski, 2024).

We used an alpha level of 0.05 for all statistical tests. However, our study was exploratory rather than hypothesis-driven, and correspondingly, we caution against over-interpretation of binary tests of statistical significance (*sensu* Forstmeier et al., 2017). We report effect sizes and confidence intervals where appropriate.

We calculated summary statistics (mean, standard deviation, maximum and minimum value, median, and interquartile range (IQR)) for bee genus richness, bee genus abundance, bee diversity (${}^1D_{bees}$), plant diversity (${}^1D_{plants}$), garden area, and urbanization index. Additionally, we quantified the number of individuals observed per genus and the occupancy of each genus (the number of sites each genus was recorded at).

We generated a genus accumulation curve to provide an assessment of our bee sampling efficacy. This was done using the *specaccum* function in the *vegan* package, and we used the option to add sites in random order.

To analyze alpha diversity (genus richness) in relation to plant diversity (${}^1D_{plants}$), garden size (log-transformed), and degree of urbanization, we initially attempted a generalized linear model (GLM) with a Poisson link. Residual diagnostics, implemented using the *DHARMa* package (version 0.4.7; Hartig, 2024), showed that model assumptions were not met. We then modeled the square-root of genus richness using a linear model (LM) with the *lm* function, and residual diagnostics revealed this to be suitable.

To analyze total bee abundance per site (total number of individuals) in relation to the same three predictors, we first attempted a GLM with a Poisson link, but this did not meet model assumptions. After log-transforming abundance values, we determined that a linear model did meet assumptions.

Using the *DHARMa* package, we found no evidence of spatial autocorrelation in the residuals for either the richness or abundance models. Using the *car* package (version 3.1-3; Fox and Weisberg 2019) we confirmed that all variance inflation factors were less than 2 for each of the richness and abundance models.

To answer the question of how the composition of native bee genera varied in relation to plant diversity ($^{1}D_{plants}$), garden size, and degree of urbanization, we undertook two complementary analyses. First, we modeled the abundances of all genera simultaneously using the *manyglm* function from the *mvabund* package (version 4.2.1; Wang et al., 2022). We used a negative binomial link, and residual diagnostics confirmed this to be suitable. We used the *anova.manyglm* function to test community-level responses to predictors.

Second, we conducted a redundancy analysis (RDA), which is a constrained ordination technique that identifies linear relationships between multivariate response data (bee genera composition) and explanatory variables. Using the *rda* function in the *vegan* package, we implemented two RDAs: one including all taxa, and one in which rare taxa (occurring in fewer than 5 sites) were excluded. Bee genus abundance was Hellinger transformed (*decostand* function in the *vegan* package) prior to running the RDAs. To test the significance of our explanatory variables, we used a permutation test for RDAs where significance for each term was sequentially assessed (*anova.cca* function in the *vegan* package, using the “by = term” option; 999 permutations). We then used the *varpart* function to calculate the percentage of the total variance in genus composition accounted for by the constrained model, and the unique percentages accounted for by individual predictors that emerged as significant. These values are akin to effect sizes.

We conducted a sensitivity analysis to explore how the results of the preceding analyses varied when alternative buffer distances were used to calculate the urbanization index. We used distances of 100m, 200m, 300m, 500m, and 700m, and excluded sites that were more proximate to each other than the given buffer distance; the resulting sample sizes were 32, 32, 30, 28, and 25, respectively. We note that the sample sizes for the 500m and 700m analyses were smaller than recommended for models with 3 independent variables (i.e. for our richness and abundance models). Below, we focus on the results corresponding to the 300 m distance (in this case including all 32 sites), and report the results of the sensitivity analyses at the end of the results section.

Results

Garden area on average was 13.5 ± 36.2 m^2 (standard error) and ranged between 3.31 m^2 and 192 m^2. On average, just under half of the plant species (47%) within an ambassador garden were species recommended by the Border Free Bees program (Fig. S4). Most of the 32 study gardens contained fewer than 20 total plant species overall, with only three gardens containing more than 30 species.

The bee genus accumulation curve (Fig. 2) indicated that our sampling effort was reasonably effective, as the rate of accumulation of new genera was generally slow after twenty sites sampled. Across all 32 gardens, we observed a total of 1871 native bee individuals, comprising 19 genera (Table S1). *Halictus* was the most abundant native bee genus (Fig. 3A), accounting for more than 35% (668) of all native bees collected. Halictidae genera *Lasioglossum* and *Agapostemon* were the next most abundant taxa (Fig. 3A). Sixteen other taxa were comparatively rarer, including 6 that were represented by 9 individuals or fewer (Fig. 3A).

Halictus individuals were collected at all 32 sites, and *Lasioglossum* were found at all but one site (Fig. 3B). Ten genera were found at 9 or fewer sites (Fig. 3B).

Neither the genus richness (Table 1) nor total abundance (Table 2) of native bee individuals within gardens varied systematically in relation to garden area, plant diversity ($^1D_{plants}$), or the degree of urbanization. Similarly, community-wide variation in the abundances of bee genera was not associated with any of the predictors based on our multivariate analysis of genus abundances (Table 3).

Within the RDA that included all bee genera (Fig. 4A), the predictors collectively explained 8.7% of the among-site variance in genus composition and revealed a significant effect of urbanization ($F = 1.85$, $P = 0.022$), which uniquely accounted for 5.8% of total community variance. Within the RDA that excluded rare genera (Fig. 4B), the predictors collectively explained 10.3% of the variance in genus composition, and revealed a significant effect of urbanization ($F = 2.56$, $P = 0.004$), which uniquely accounted for 6.8% of total community variance. The effect of urbanization in each model remained significant after adjusting for multiple testing (Bonferroni adjusted alpha = 0.05/2 = 0.025).

The biplots (Fig. 4) indicate that the majority of the rare genera are situated near the origin (Fig. 4A), indicating minimal correlation with our explanatory variables. Among the less rare genera, *Heriades* exhibited a positive association with both plant diversity ($^1D_{plants}$) and garden area while *Agapostemon* exhibited a positive association with urbanization (Fig. 4). In contrast, the most abundant genus *Halictus* exhibited a somewhat negative association with urbanization.

The results of the preceding four statistical analyses remained qualitatively similar when using different buffer distances for calculating the urbanization index. Specifically, (1) the richness and abundance regressions were consistently non-significant (i.e. the 95% confidence intervals for all predictors' standardized coefficients consistently overlapped zero; Fig. S5); (2) the multivariate analysis of genus abundances consistently yielded non-significant predictor coefficients (Fig. S6); and (3) urbanization was in most instances the sole predictor to account for a significant (Fig. S7) but small percent (Fig. S8) of the among-site variance in community composition (with all genera included, ranging from 1.0 to 6.1% from smallest to largest buffer distance, and 1.2 to 7.3% when rare genera were excluded).

Discussion

A primary goal of this study was to quantify, for the first time, bee diversity among bee-friendly ("Bee Ambassador") urban gardens located within the Okanagan diversity hotspot of British Columbia, Canada. We also aimed to quantify the responses of native bee communities to the local and landscape-scale characteristics of the gardens. We observed almost 50% of the bee genera found in the biologically diverse Western Interior Basin region of British Columbia (Sheffield and Heron, 2018). Many of the genera we did not observe during our study are considered uncommon, including *Calliopsis*, *Epeolus*, and *Xeromelecta* (Sheffield and Heron, 2018). Others, such as *Habropoda* (Barthell et al., 1998) and *Anthophora* (Stone, 1993), typically experience their period of activity during the spring months, which were missed by our sampling efforts that started in mid-June.

Overall, we found that none of the local garden characteristics nor the degree of urbanization had a detectable influence on native bee genus richness or total abundance of individuals. Moreover, although urbanization accounted for a statistically significant percentage of total community variance among gardens, the percentage was rather small (5.8% when all bees were included and 6.8% when rare genera were excluded). Nevertheless, this does suggest that some bee genera may respond differentially to urbanization.

Previous work suggests that landscape-level factors do not always play a large role in predicting the abundance and richness of bees within urban environments (Quistberg et al., 2016; Ayers and Rehan, 2021; Rotondi et al., 2024). Our findings remain consistent with previous studies, which include a mixture of results relating richness to urbanization, ranging from positive influence (Martins et al., 2017; Cohen et al., 2022) to negative influence (Simao et al., 2018; Birdshire et al., 2020; Villalta et al., 2022), to relatively no influence (Banaszak-Cibicka and Żmihorski, 2012) of increased levels of urbanization on bee richness and abundance. Our results most similarly relate to those of Banaszak-Cibicka and Żmihorski (2012) and Fowler (2014), who found relatively little variation along an urban gradient. The differences in response of bee communities to urbanization may also be related to how studies have classified urbanized areas. The studies that found a negative response (except for Villalta et al., 2022), used study sites with high impervious cover. For example, Birdshire et al., (2020) classified urban sites if they had at least 50% impervious cover and Simao et al., (2018) purposely chose sites that bordered paved parking lots. The lack of any clear pattern in our data indicates that urbanization is not having an impact on overall bee abundance and richness in urban gardens within our study system. This may be due to genus specific differences in response to urbanization, regional differences in impervious surface cover, garden area, and diversity of plants.

The RDA biplots (Fig. 4) suggest some genus-specific responses to urbanization, garden area, and plant diversity in our study gardens, similar to other studies showing that pollinators from different taxonomic groups respond differently to these factors (Neumann et al., 2023; Brasil et al., 2024). However, conflicting results have been reported among studies examining the impacts of urbanization on bees with different traits (Buchholz and Egerer, 2020). Cavity nesters such as *Megachile* may benefit from the increased impervious surfaces in more urbanized areas compared to ground nesters (Felderhoff et al., 2023). However, Brasil et al. (2024) found ground nesters to have a positive association with urbanization. We also observed a conflicting relationship with urbanization and ground nesters with some showing a positive relationship (*Agapostemon, Eucera, Melissodes*) and others a negative relationship (*Andrena, Pseudopanurgus,* some *Bombus*). Some of these differences may be due to interacting effects. For example, increasing plant diversity, especially of host or preferred floral resources of more specialized species within the genera (i.e., *Melissodes, Eucrea, Andrena*) may offset some negative impacts of urbanization (Burdine et al., 2019; Birdshire et al., 2020). Additionally, other factors not measured in our study might strongly determine the presence of genera, such as whether the host species for cuckoo genera (i.e., *Sphecodes*) are present. Other research focused on urban gardens and different gardening practices, such as promoting nesting habitat by purposefully leaving bare ground or logs, could mitigate the impacts of urbanization, while planting flowering trees can particularly benefit early-emerging bee species (Splitt et al., 2021). These potential garden features were not measured in our study, but may have had an impact on bee genera composition among our study sites. Overall, these findings highlight the need for nuanced conservation strategies that consider species-specific traits, interactions among garden features, and local management practices to support diverse pollinator communities in urban areas.

Among the local-scale variables, we found that there was very little effect shown for bee genera richness. Sites with greater floral diversity should generally foster more diverse bee populations (Theodorou et al., 2020; Gerner and Sargent, 2022), although this was not the case within our study. Increased garden area is typically positively associated with bee richness and pollinator diversity overall (Shwartz et al., 2013), however garden area had no significant association with richness. Bee richness and total abundance may not have responded to increases in plant diversity or garden area because other local scale factors that we did not measure may have been more important drivers. These could include bees selecting gardens based on certain forage species (Wojcik and McBride, 2012; Dylewski et al., 2020; Sirohi et al., 2022), particularly high-quality forage (Somme et al., 2015; Fowler et al., 2016; Ruedenauer et al., 2016), or availability of nesting resources which may be missing from urban gardens through maintenance and gardener preferences (Antoine and Forrest, 2021; Pfeiffer et al., 2023). This may especially be the case for our rarely sampled bees which include bare-ground, formed nests out of rock and dirt, and pithy stems. Additionally, our cuckoo bees would have been limited by the presence of their host species and not garden characteristics. Limited statistical power may have also constrained our ability to detect the effects of local garden characteristics, due to small sample size and/or because the true effect sizes of local garden characteristics were too small. Our bee sampling methods may also have been limiting; while the netting of specimens is a popular collection technique used in many similar studies, even alongside the usage of pan traps (Grundel et al., 2011; Quistberg et al., 2016) to accommodate for sampling biases, it was not feasible in this study. The use of netting techniques can be fairly destructive to vegetation and since the study used private home gardens, there was a risk of damage. A major limitation of pan traps is their ineffectiveness at capturing larger specimens, such as bumble bees (Prendergast et al., 2020). Ultimately, a combination of these two methods could provide a more accurate representation of the overall bee community within a garden (Quistberg et al., 2016).

The dominance of the Halictidae family on the abundance data of observed bee specimens is worth noting. The three most abundant genera (*Halictus*, *Lasioglossum* and

Agapostemon) all belong to this family, accounting for a combined 66.4% (1243 individuals), approximately two-thirds of all 1871 native specimens collected in this study. Individually, *Halictus* accounted for more than one-third (35.7%) of these specimens. It should come as little surprise that genera of this family were overwhelmingly dominant, as it is an extremely abundant family in North America and is especially abundant among pan trap collections (Hinners et al., 2012). As with general abundance, the Halictidae family dominated genus composition across sites. In fact, individuals of the genus *Halictus* appeared at all 32 sites, with *Lasioglossum* present at all except one. The dominance of Halictidae, and *Halictus* in particular, across all sites may help explain the limited amount of variation accounted for by the RDA models.

Notably, all our findings remained qualitatively consistent when we used different buffer distances for quantifying urbanization (Figures S5-S8). Previous studies examining landscape effects on bee communities have used a variety of buffer distances. Some urban studies have focused on similar, and smaller distances (Johansson et al 2018), or larger distances (Turo et al 2021) while others have used a wider range than we employed here (Davis et al 2017; Rajbhandari et al 2023; Landsman et al 2024). The distance at which landscape effects were most significant for bee communities varied widely. For example: in urban areas of the northeastern U.S., a 1000 m distance was most informative for bee community structure (Landsman et al 2024); in New York City, U.S.A., a 500 m distance best explained bee visitation; in Cleveland, U.S.A., higher bee richness and abundance was observed at the 1500 m distance (Turo et al 2021); While in Chicago, U.S.A. (Davis et al 2017), and in Sollentuna, Sweden (Johannson et al 2018), 250 m and 200 m were the most informative, respectively. The range of buffer distances selected for our sensitivity analysis is consistent with those used in other research on urban bee communities.

One limitation of our study was that the sampling occurred in only a single year. Inter-annual variation in bee community composition and abundance can make it difficult to make definitive conclusions particularly for long-term trends, but other studies have found that while inter-annual variation exists, variation in bee communities is much larger between months (Turley et al 2022). Indeed, when there are large interannual shifts in community composition, when temperatures are warmer than average for example, it is likely that all bees will shift in phenology in a similar manner as has been observed for butterflies (Gutiérrez and Wilson 2021). Therefore, while it is important to capture inter-annual variation, it may be more crucial to sample as much of the active period for bees as possible. Moreover, when there is inter-annual variation observed, the most dominant species tend to be relatively stable (Senapathi et al 2021). This may mean our results would remain consistent between years for the most commonly observed taxa. However, previous studies have noted significant differences in interactions among species between years (Chacoff et al 2018; Prendergast et al 2021) which may impact our results if different garden plants attract different bees or the presence of other species has a varying effect between years. Future research should incorporate multi-year sampling to validate our findings and provide a more comprehensive understanding of the long-term impacts of urban gardens on bee communities.

This study was the first to obtain estimates of bee diversity within urban gardens in the Okanagan. In doing so, it explored potential relationships between bee diversity and factors that have previously been hypothesized to be influential, including plant diversity, garden size, and degree of urbanization. We could not evaluate the efficacy of the Bee Ambassador program for enhancing bee diversity, as we were unable to secure access to appropriate “control” gardens in time for the study. Residential gardens containing larger proportions of native plants may see increased levels of pollinator activity and as programs and initiatives such as Border Free Bees heavily promote the planting of native species, the effect this has on the attraction of bees is little understood (Pardee and Philpott, 2014). We were unable to incorporate this into our analyses for logistical reasons, however, it should be examined in any similar future studies within the Okanagan. Thus, there remain no studies examining the effectiveness of this type of initiative on the conservation of native urban bee populations. A larger geographic scope and longer-term approach to this type of study could be beneficial in truly understanding the dynamics of native urban bee communities within Kelowna and the potential effects of such pollinator gardens. Being the first study to systematically estimate bee diversity among urban gardens in Kelowna, future studies should build off our findings by using a number of control sites in addition to main study gardens to allow for effective comparison between average gardens and those deemed pollinator-friendly. In doing so, more accurate measures of the true effects of pollinator gardens on bee community dynamics can be obtained. Although the Bee Ambassador Project has ended, its efforts to raise awareness about the importance of native pollinator gardens should continue. Future urban pollinator garden efforts may benefit from providing resources guiding plant species selection, and engage in efforts to recruit a diversity of community members to help create an extensive network of pollinator-friendly habitat throughout their city.

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Figures

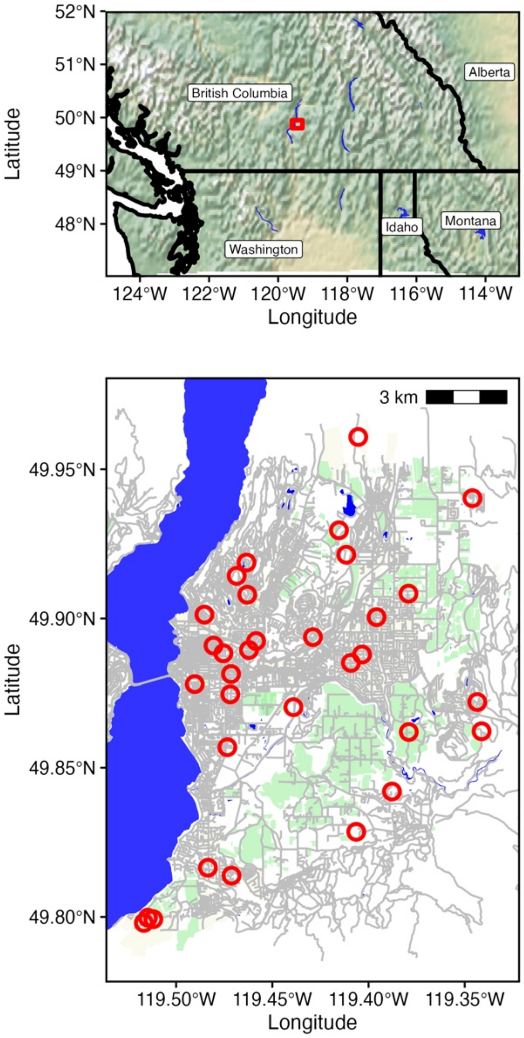

Fig. 1: Approximate location of the 32 gardens (red circles) within the city of Kelowna and environs (bottom), and a reference map (top) showing the location of the city of Kelowna (red box) within British Columbia. The top map is unprojected (WGS84 datum) and the map of Kelowna uses UTM Zone 11N (NAD83).

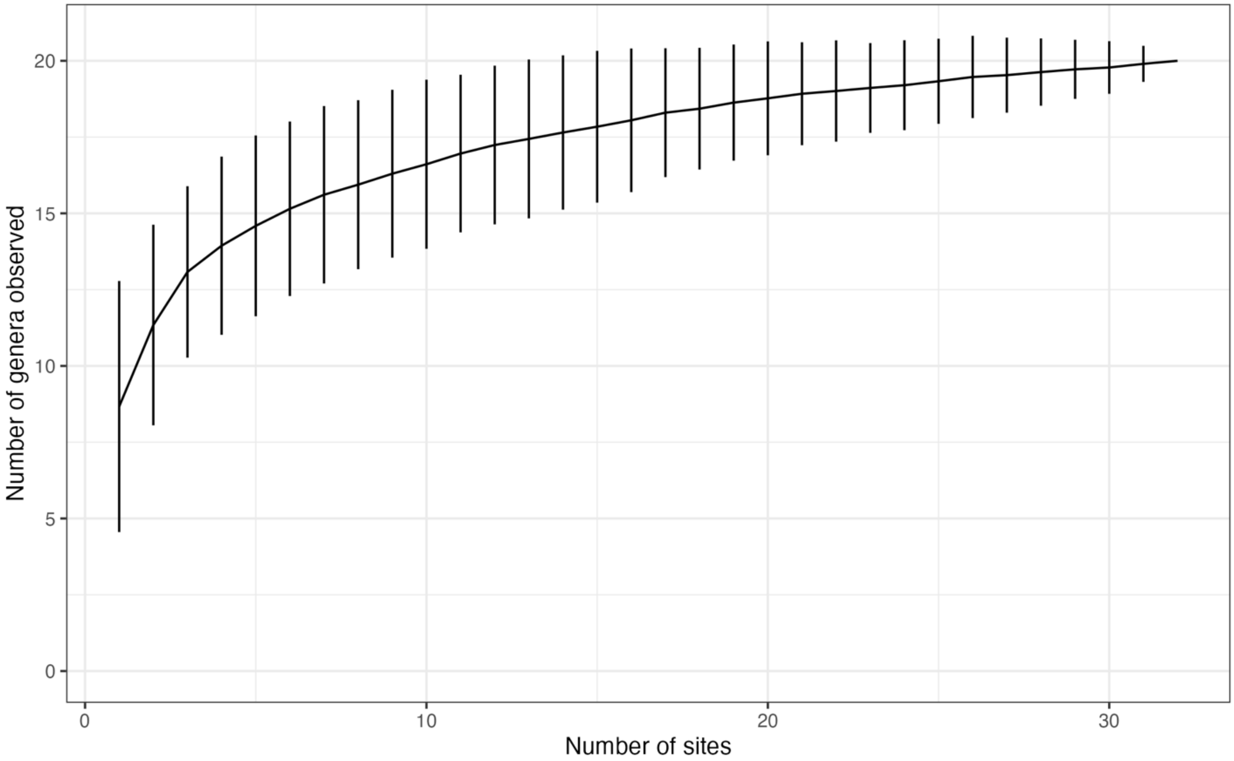

Fig. 2: Genus accumulation curve. Error bars extend to 1.96 x the standard error above and below the randomization-based estimate of mean genus richness (curve) for increasing number of sites (maximum N = 32) sampled in Kelowna, British Columbia between June 10 and August 29, 2019.

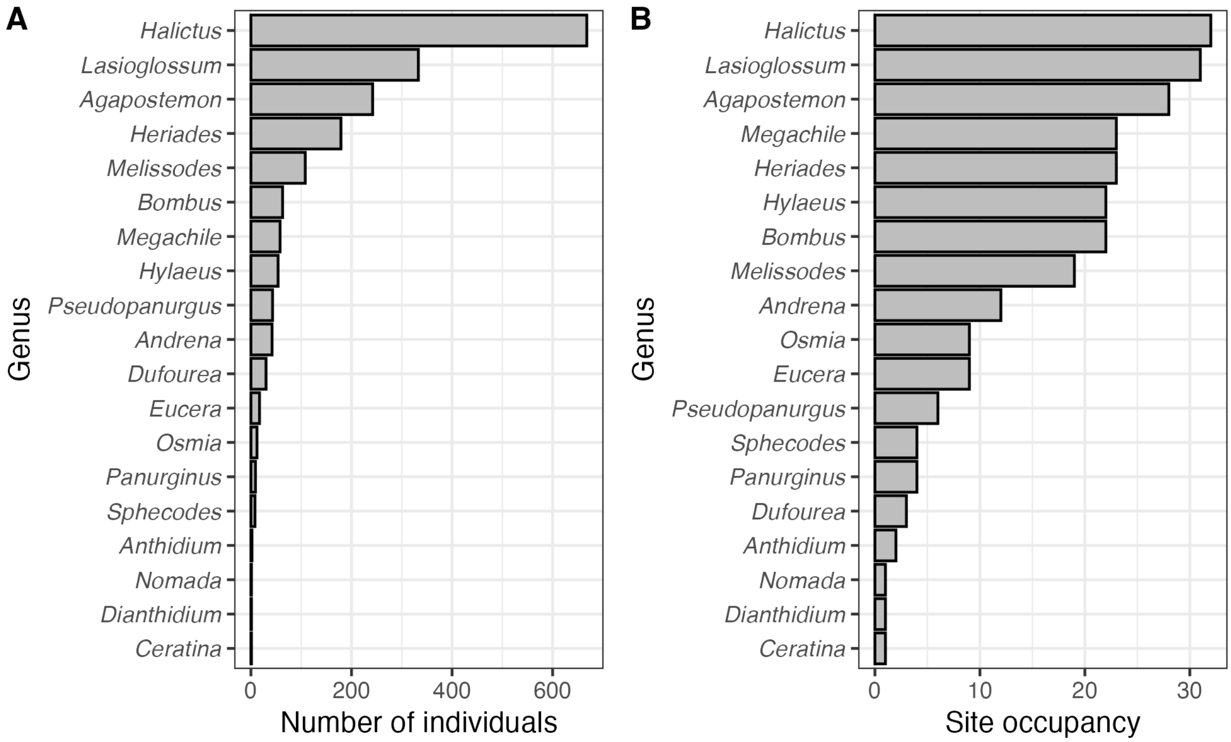

Fig 3: Barplots of the abundance (A) and occupancy (B) of 19 native bee genera observed across 32 gardens in Kelowna, British Columbia, sampled between June 10 and August 29, 2019.

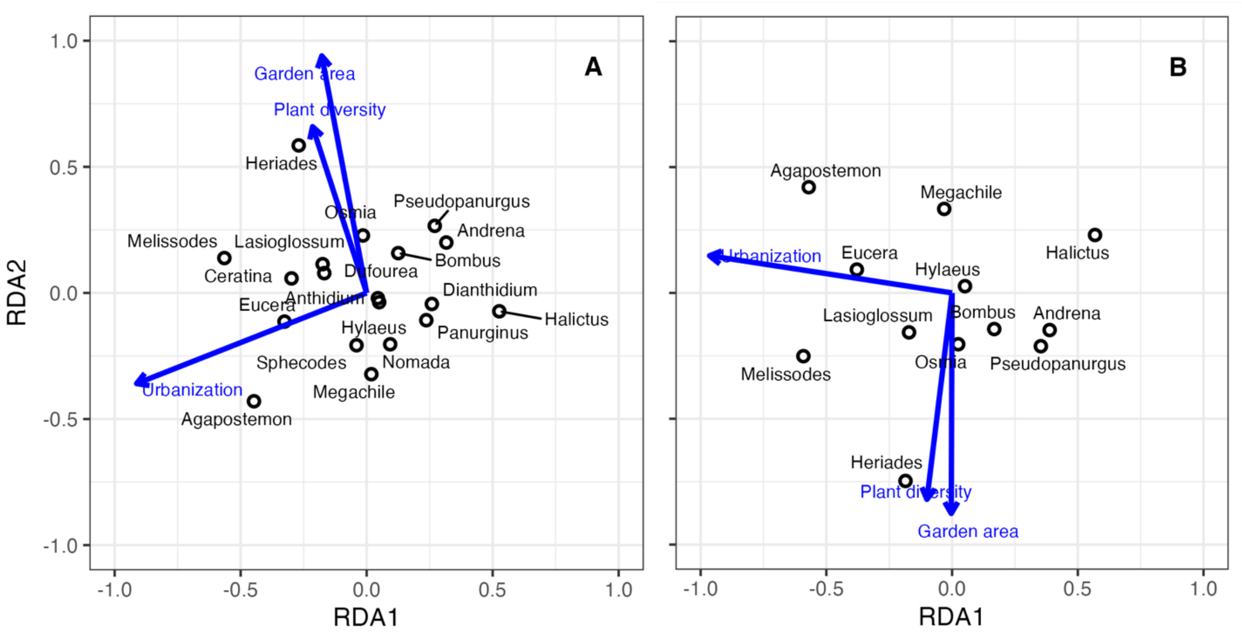

Fig. 4: Biplots from RDAs evaluating how the composition of (A) all bee genera and (B) with rare genera removed varies in relation to urbanization, plant diversity (${}^{1}D_{plants}$) and garden area across all sites. In panel (A), axis RDA1 accounted for 6% of the variance in bee genera composition and RDA2 accounted for 4% (total inertia: 19). In panel (B) axis RDA1 accounted for 7.8% of the variance and RDA2 accounted for 5.8% of the variance (total inertia: 12). Bee sampling was conducted across 32 gardens in Kelowna, British Columbia, between June 10 and August 29, 2019.

Supplemental Tables and Figures for:

Native bee genus diversity within bee-friendly urban gardens varies little along an urbanization gradient

Table S1: Descriptive statistics for variables quantified at each of the 32 gardens in Kelowna, BC in 2019. SD = standard deviation, IQR = inter-quartile range.

	bee genus richness (# unique genera)	total bee abundance (# individuals)	bee genus diversity ($^1D_{bees}$)	plant diversity ($^1D_{plants}$)	garden area (m^2)	Urbanization (within 300m)
mean	7.88	58.5	4.93	10.00	13.5	0.00
SD	2.21	43.0	1.18	5.52	36.2	1.29
minimum	4.00	7.00	2.96	1.65	3.31	-2.05
maximum	12.0	173	6.96	24.64	192	2.74
median	8.00	50.0	4.82	8.61	16.5	0.086
IQR	3.25	43.3	1.93	5.90	20.7	1.94
Lower 95% CL	7.11	43.56	4.52	8.18	18.94	-4.47
Upper 95% CL	8.64	73.38	5.34	11.82	44.00	4.47

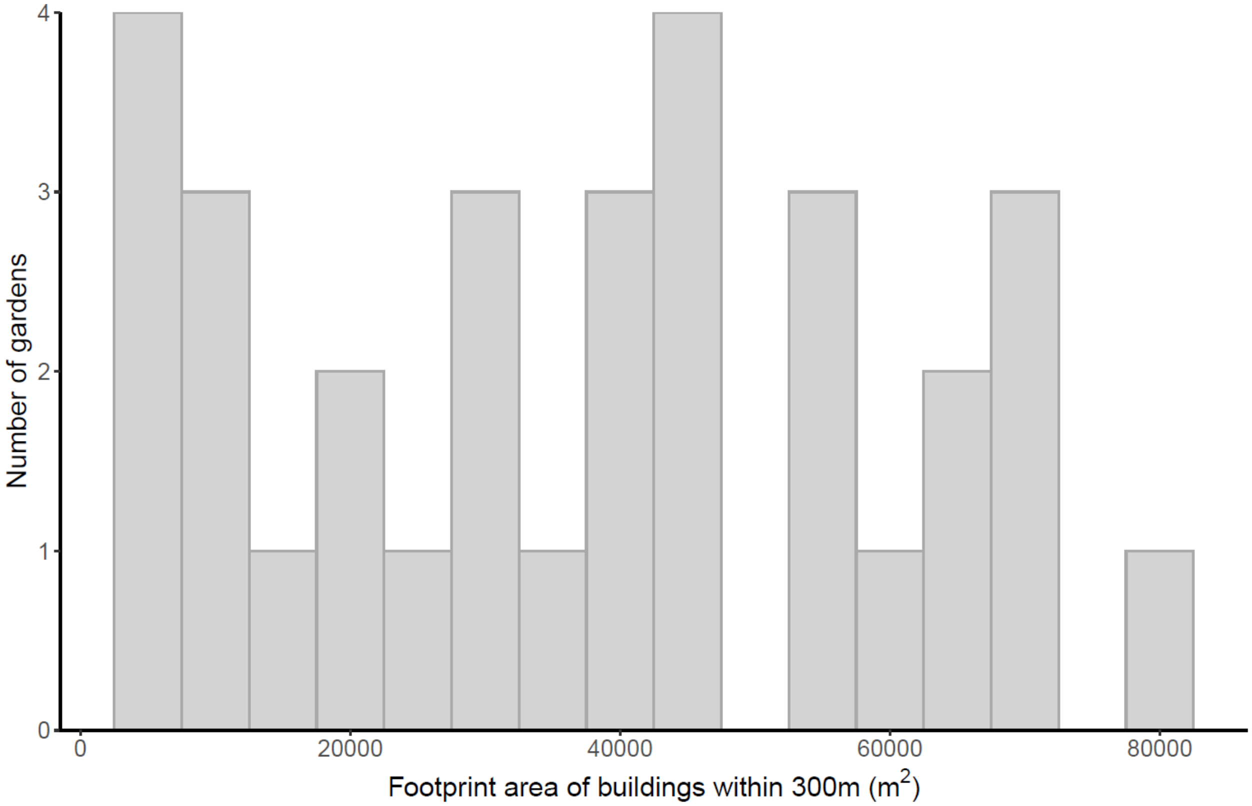

Fig. S1: Histogram showing the total footprint area of buildings (m^2) within 300 m of each of the 32 sampled gardens.

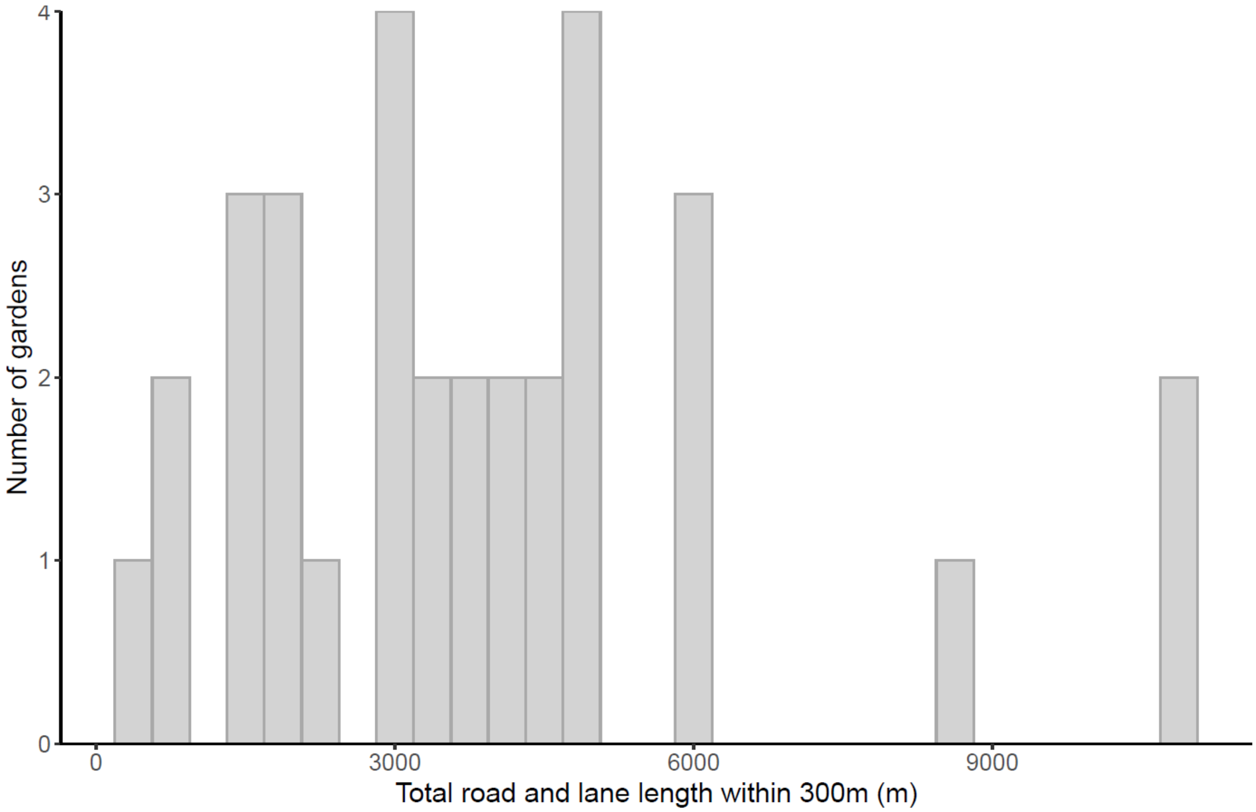

Fig. S2: Histogram showing the total road and lane length (m) within 300 m for each of the 32 sampled gardens.

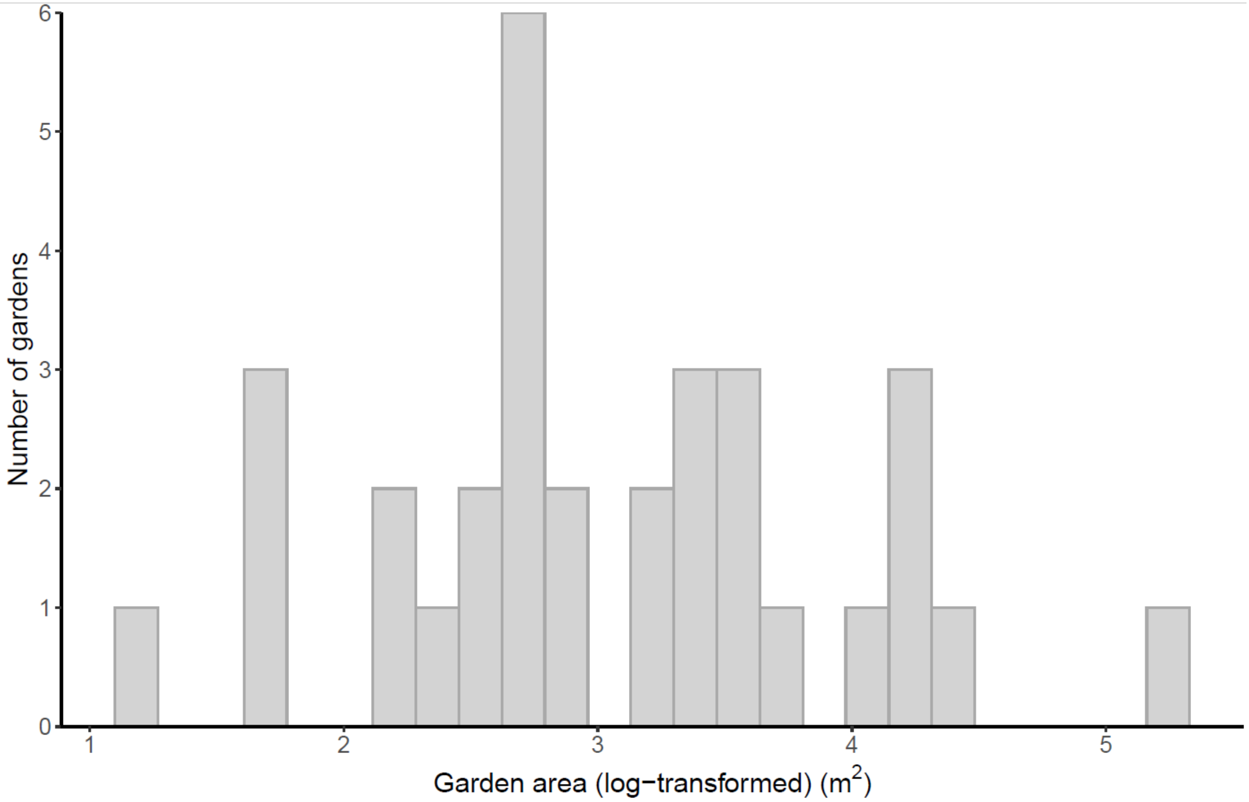

Fig. S3: Histogram showing garden area (log-transformed) for each of the 32 gardens in our study.

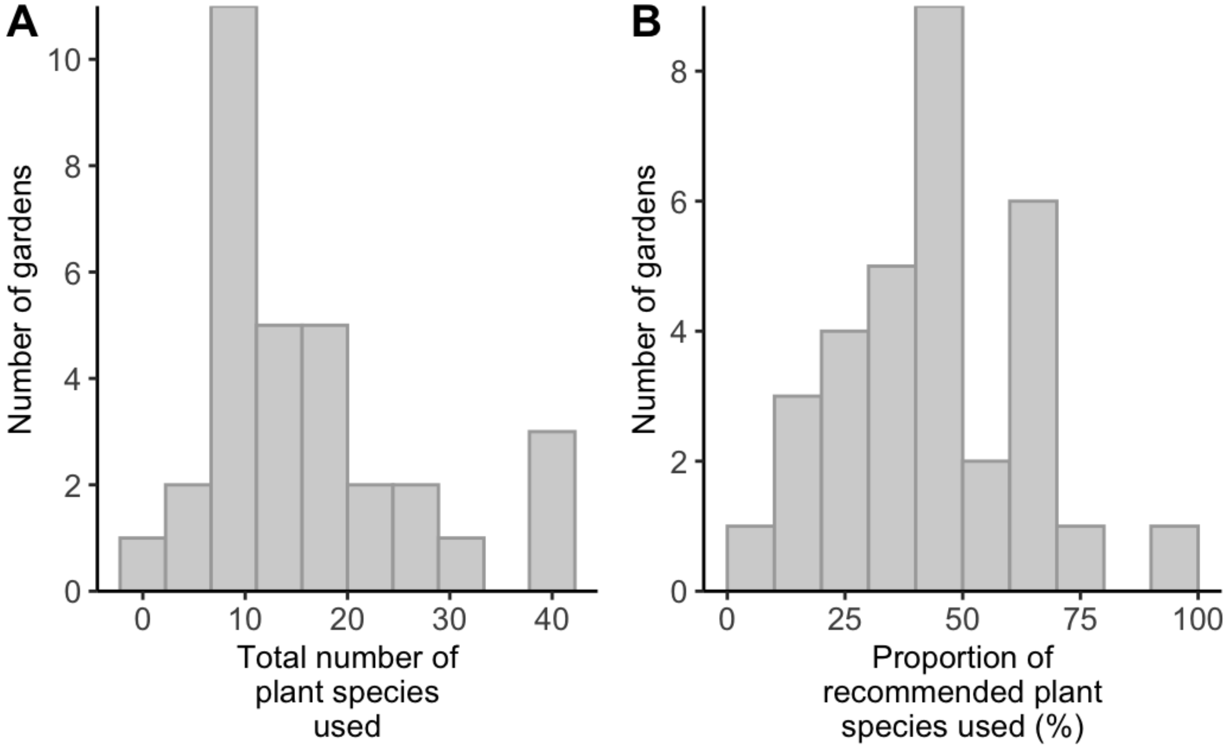

Fig. S4: (A) Histograms showing (A) the total number of plant species used and (B) the the proportion of those plant species listed as recommended species by Border Free Bees. Sample size: N = 32 gardens.

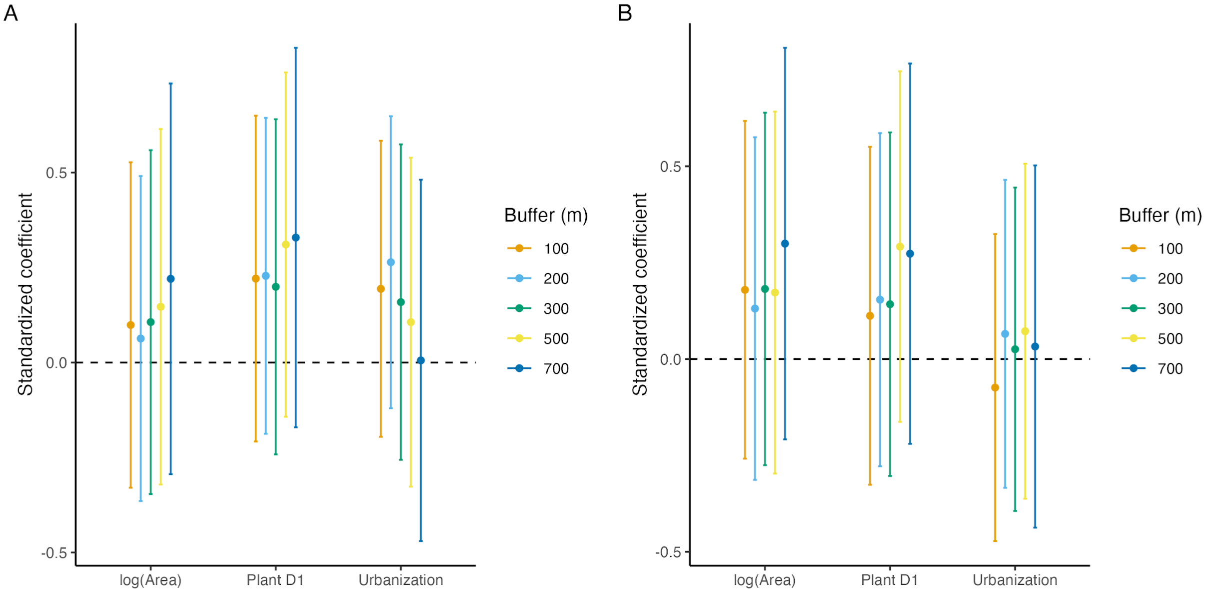

Fig. S5: Standardized coefficients (solid points) and their respective 95% confidence intervals (segments) for each of the three predictors (garden area, log-transformed; plant diversity (${}^1D_{plants}$), and urbanization index) within linear models using (A) genus richness and (B) total bee abundance per garden as response variables. Different colours indicate the different buffer distances used to calculate the urbanization index. Sample sizes for both regressions at each of the buffer distances (from smallest distance to largest) was N = 32, 32, 30, 28, and 25, respectively.

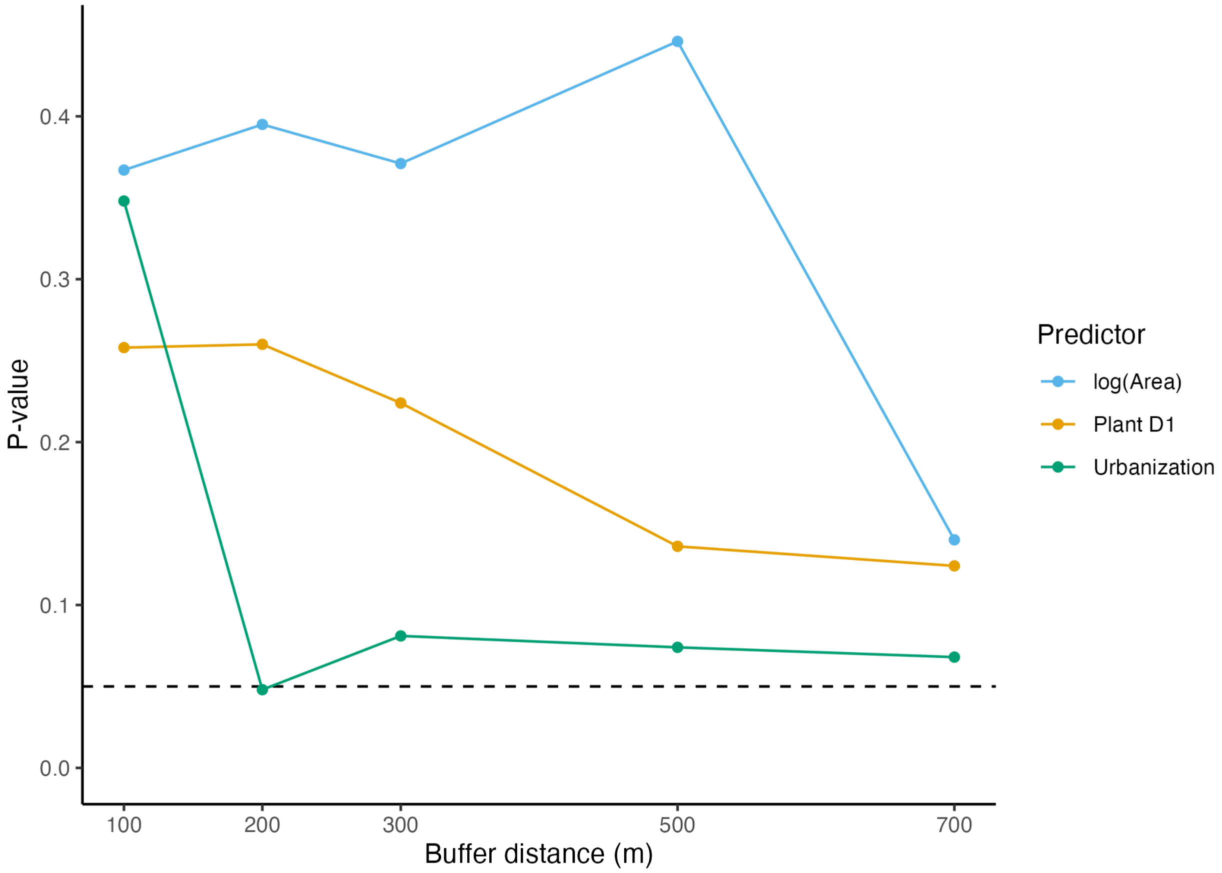

Fig. S6: P-values for each of the three predictors (garden area, log-transformed; plant diversity ($^1D_{plants}$), and urbanization index) within the multivariate generalized linear model, plotted against the different buffer distances used to calculate the urbanization index. Sample size for the regression at each of the buffer distances (from smallest distance to largest) was N = 32, 32, 30, 28, and 25, respectively.

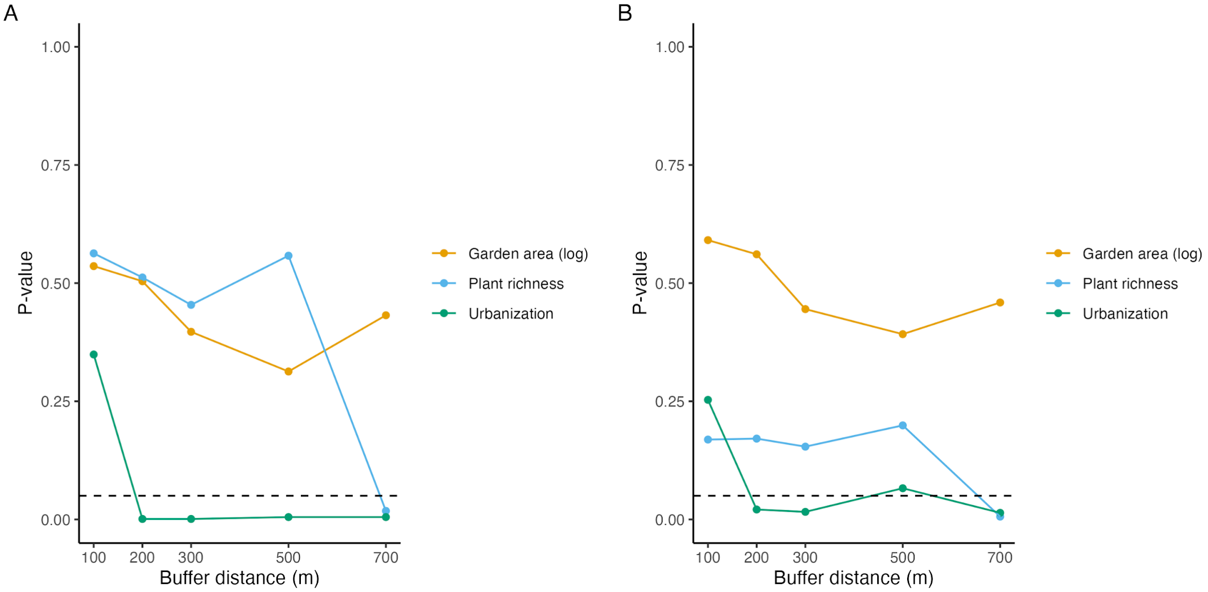

Fig. S7: P-values for each of the three predictors (garden area, log-transformed; plant diversity ($^{1}D_{plants}$), and urbanization index) within the redundancy analyses (RDAs), plotted against the different buffer distances used to calculate the urbanization index. Shown are the results for the RDAs that (A) include all bee genera and (B) exclude rare bee genera. Sample size for the RDA at each of the buffer distances (from smallest distance to largest) was N = 32, 32, 30, 28, and 25, respectively.

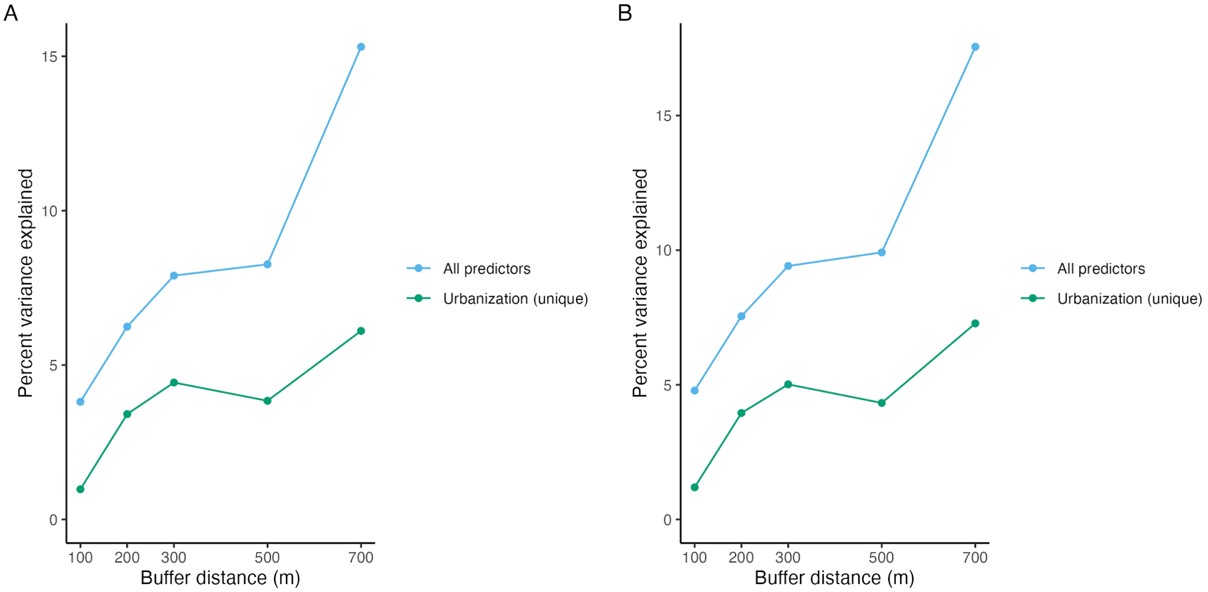

Fig. S8: Percent of community variance explained by all three predictors combined (garden area, log-transformed; plant diversity ($^1D_{plants}$), and urbanization index), and uniquely urbanization (green line) within the redundancy analyses (RDAs), plotted against the different buffer distances used to calculate the urbanization index. Sample size for the RDA at each of the buffer distances (from smallest distance to largest) was N = 32, 32, 30, 28, and 25, respectively.