The effects of impervious surface area, tree canopy cover, and floral richness on bee abundance, richness, and diversity across an urban landscape

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Table of Contents

List of Tables

List of Figures

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Abstract

As urbanization increases globally, habitat loss is increasing at an unprecedented rate, eroding the suitability of many landscapes for most forms of wildlife, including bees. At least some of this habitat loss is through the ongoing expansion of urban areas, a process termed 'urbanization'. Studies of the effects of urbanization and urban land use on bees have reported a mixture of results, including some instances where at least some species appear to do better in urban areas than they do in lands surrounding urban areas. While the impacts of urbanization on bee communities has been investigated, tree canopy cover has been largely overlooked as a contributor to urban bee distributions, despite their potential importance as a predictor of bee activity. I investigated the impacts of urban land use and tree canopy cover on bee communities across a variety of neighbourhoods in a medium-sized Canadian city (Ottawa, Ontario). In total, I surveyed bee communities in 27 residential yards that varied in terms of the degree of urban land use (measured as percent impervious surface area) and tree canopy cover (percent deciduous canopy cover) across a range of spatial scales. Using linear regression and model selection, I determined that bee abundance was negatively correlated with the degree of urban land use surrounding a yard, and positively correlated with the richness of the local (i.e., yard) flowering plant community. Yard floral richness, but not urban land use or tree cover, was also a predictor of the diversity of the bee community visiting the yard. In fact, tree canopy cover did not factor in any of the top models predicting either bee abundance, richness, or diversity. My results support the idea that urban land use could negatively impact bee communities, but also suggest that landscaping and urban planning decisions that maximize

local floral richness could counteract some of the negative impacts of urbanization on bee populations.

Chapter 1: **Overview**

Over 55% of the global population currently lives in an urban area; by 2050 this is expected to rise to 68% as the population continues to grow (United Nations, 2018). The conversion of natural lands to urban use is a leading contributor to habitat loss (Vanbergen, 2013; Winfree et al., 2009), and is associated with species decline (Czech, 2000). Urbanization typically alters existing landscape to include more patches of impervious surface area (McKinney, 2002) with the remaining land focused on human-use, and few considerations for wildlife (Müller et al., 2013). However, the inclusion of green spaces such as gardens, parks, and wilderness areas may help to mitigate the negative impacts of urbanization on wildlife (Hunter & Hunter, 2008), which may be particularly true for small, mobile organisms such as insects (Torné-Noguera et al., 2014).

Urbanization is the process of human population and economic growth that results in the expansion of impervious surfaces, and a corresponding reduction of natural habitat (Bertinelli & Black, 2002). Impervious surfaces include roads, houses, buildings and other concrete or asphalt structures (Yan et al., 2019). These surfaces are commonly used to represent urbanization in studies investigating bee abundance (Hamblin et al., 2018; Sivakoff et al., 2018), richness (Threfall et al.,2015), and diversity (Ahrne et al., 2009; Fortel et al., 2014; Glaum et al., 2017) in urban environments. Impervious surface cover estimates can provide a quantitative framework to help evaluate the effects of urbanization (Zhang et al., 2012). Accordingly, I used impervious surface area as a proxy for urbanization; going forward, mentions of urbanization refers to the percentage of impervious surface in a defined area.

The theory of island biogeography (IBT, MacArthur & Wilson, 1967) is a hypothesis to explain how the number of species on an island are impacted by the size of the island and its isolation (distance) from the mainland. The theory has also been co-opted to try to understand how fragmented landscapes, including those affected by urbanization, impact species abundance and diversity. In the analogy, habitat fragments are considered akin to 'islands' of habitat (Brown, 2008), surrounded by an inhospitable matrix (e.g., buildings, roads), where connectivity between patches of suitable habitat is weak (Torné-Noguera et al., 2014). According to IBT, islands (fragments) that are smaller and less connected tend to support smaller populations/receive less migration and are more extinction-prone (MacArthur & Wilson, 1967). In an urban setting, habitat quality for many organisms tends to decline along a gradient from the outlying regions towards the city core (McKinney, 2002). While IBT has previously been used to predict species diversity across urban areas (Fattorini, 2014), it may have limited predictive power for insect communities, possibly due to fragment connectivity that is unlike what occurs on islands (Fattorini et al., 2018).

Urban areas are also often associated with the degradation of critical habitat to many endemic and at-risk species (Elmqvist et al., 2014). For example, the highly urbanized San Francisco Bay Area has one of the highest population densities in California (U.S. Census, 2010) and is also a hotspot for endangered arthropods (Connor et al., 2002). It has been estimated that 43% of the Bay Area's butterfly species have been lost to habitat destruction (Hafernik & Reinhard 1995). Indeed, three of the Bay Area's endemic butterfly species are already extinct (Connors et al., 2002). This is also true for the Bay Area's bee fauna, where recent surveys of urban and wild parks found only three of the nine bumblebee species historically collected in

these areas (McFrederick and LeBuhn, 2006). Closer to home, southwestern Ontario's unique Carolinian zone is a biodiversity hotspot, supporting many rare and endemic species, including 542 species of rare vascular plants (Klinkenberg, 2002). Comprehensive insect surveys have not been performed, but Ontario has the second-highest number of bee genera in Canada after British Columbia, with approximately 400 wild bee species (Ascher & Pickering, 2020). Unfortunately, southern Ontario, including the endangered Carolinian zone, is among the highest population density areas in Canada (it contained 33% of Canada's population in 2016), even though it accounts for only one percent of Canada's total land mass (Government of Ontario, 2019), putting many endemic species at risk due to increasing urbanization.

Despite these negatives, urban areas can foster relatively high levels of biodiversity (Baldock et al., 2019; Cornelis & Hermy, 2004; Gaston et al., 2005). Management of private gardens and parks is associated with high ecosystem diversity as inputs like fertilizers and a diverse array of plant species increase productivity (McKinney, 2008). Urban areas also tend to host a higher proportion of exotic plant species, and consequently, a higher overall plant diversity than surrounding lands (McKinney, 2006). Similarly, cities can be hotspots for species that are urban specialists or those that survive by taking advantage of human-made structures or processes (Desaegher et al., 2018). While this increases urban diversity, it also reduces the uniqueness of species compositions among urban environments (Baldock et al., 2015; McKinney, 2002; Shochat et al., 2006) by "ecologically filtering" which species persist (Harrison & Winfree, 2015).

High habitat quality, typically defined as habitat that increases the fitness of an individual (Van Horne, 1983), is positively associated with bee richness (Hopfenmüller et al.,

2014). Bees require floral resources (i.e., pollen and nectar) and adequate nesting substrate to successfully rear their young (e.g., bare soil, cavities, or plants with pithy stems) (Potts et al., 2010). In urban areas, the quality of these two requirements may be reduced through the degradation of soil/loss of soil/loss of natural cavities (e.g., removal of tree snags, lack of bare ground) (Alvey, 2006) and urbanization-associated plant species compositional changes (Williams et al., 2015). On the other hand, habitat quality can be increased through additional plantings of floral resources (Baldock, 2020). Thus, land management can help to mitigate the negative effects of urbanization by providing resources that bees need.

Trees are an important, but often overlooked, component of the urban landscape that can increase habitat quality for a wide variety of species (Somme et al., 2016). Urban tree cover is defined as the parts of a tree (e.g., leaves, branches, and trunk) that, when viewed aerially, cover the ground layer (USDA, 2008). In the United States alone, four million trees are lost annually to urban sprawl; while urban canopy regeneration does occur, on average, tree cover is decreasing in most major US cities (Nowak & Greenfield, 2012). In Canada, urban tree cover was estimated to be 26% in 2012, which is a 1.5% decrease from 1990 (McGovern & Pasher, 2016), suggesting that similar declines may also be occurring in a Canadian context (Nowak & Greenfield, 2012). Despite their ubiquity in the environment, few studies have looked at urban trees and their potential to negatively affect bees through the shading of understory flowering plants. Part of the problem is that tree cover is constantly changing; especially in areas with high urbanization, as trees are felled to make way for impervious surfaces (Nowak & Greenfield, 2012). High-quality tree cover data is also often rare or outdated, which adds to the difficulty in accurately assessing canopy changes over time (Richardson & Moskal, 2014).

The majority of tree canopy cover studies that investigate wild bees have looked at forested, not urban ecosystems (e.g., Grundel et al., 2010; Odanaka & Rehan, 2020; Roberts et al., 2017). In these studies, bee abundance, richness, and diversity tend to decline as tree cover increases (McCabe et al., 2019; Odanaka & Rehan, 2020; Roberts et al., 2017; Winfree et al., 2007, although see Taki et al., 2007). However, if the ecosystem is highly disturbed (e.g., through fire or management) and retains characteristics of early successional forests, bee richness (Rubene et al., 2015) and abundance (Potts et al., 2003) tend to be higher than in typical forested environments. These findings suggest that trees can affect bee population distributions; yet, studies of tree canopy cover on bees in urban ecosystems are relatively rare. A Birmingham, UK study looked at tree canopy cover across an urban gradient and found a 50% reduction in bee richness as tree cover increased (from 5-45% in a 100m radius) (Brunbjerg et al., 2018). This result is supported by another study in Toledo, Ohio, that looked at urban parks and gardens and also found reductions in bee abundance as canopy cover increased, although the authors did not measure canopy cover at a landscape scale (Burdine & McCluney, 2019). To my knowledge, no other study has focused on how patterns of bee species abundance, richness and diversity correlate with tree cover on a landscape scale in residential yards (but see Brunbjerg et al., 2018; Lowenstein et al., 2014; Matteson & Langellotto, 2010, for papers that examine tree cover in other urban greenspaces).

Urban land managers and NGO's can promote strategies that increase bee foraging and nesting resources, with the potential to increase the critical bee habitat available in urban areas (reviewed in Dicks et al., 2010). For example, 'wildlife gardening' is a practice in which the public is encouraged to aid wildlife establishment through the application of certain practices

(i.e., planting a diversity of flowering plants, reducing pesticide use, leaving areas unmanaged) (Dicks et al., 2010; Goddard et al., 2013). By applying these methods, cities can provide adequate floral and nesting resources for many bee species (Hall et al., 2017). In my thesis research, I examine the role of two major types of urban land cover, impervious surface area and tree canopy cover, as well as flowering plant species abundance and richness, on bee species' distribution and abundance across the City of Ottawa, Ontario. Through field surveys and statistical analyses, I examine the degree to which changes in these factors predict bee abundance, richness, and diversity in residential yards that span a gradient in these two types of land cover. Various studies have independently looked at the effects of urbanization (Fitch et al., 2019; Hung et al., 2017; Matteson et al., 2008) and tree canopy cover (Farwig et al., 2009; Grundel et al., 2010) on bee abundance, richness, and diversity. To my knowledge, only one study has looked at them as landscape variables in urban gardens (Quistberg et al., 2016); yet, this study only examined forested tree cover, which excludes the impacts of urban canopy cover on bee diversity measures. My thesis work helps to fill in missing knowledge about the drivers of bee abundance, richness, and diversity in an urban landscape.

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Chapter 2: Landscape and local drivers of bee diversity and abundance in urban areas

Pollination is an ecosystem service that is beneficial to the sexual reproduction of upwards of 87.5% of plant species (Ollerton et al., 2011), often through the activities of wild and managed bees. There is growing evidence that globally, wild bee species are in decline (Potts et al., 2010), with land-use change as a major driver (Baldock et al., 2019). In particular, urbanization causes widespread habitat loss by eliminating natural habitat (Jones & Leather, 2012). As urban areas continue to expand, cities must be better designed for the inclusion of wild bee habitat (Hunter & Hunter, 2008; Tanner et al., 2014).

Private gardens can serve as a refuge for urban insect populations by providing habitat and flowering plants (Hall et al., 2017). Urban gardens can provide floral resources when site owners plant ornamental and exotic flowers in their yards (Thompson et al., 2003). For example, stingless bee foraging activity (measured by counting the number of returning workers to the hive) in Queensland, Australia was found to be higher in suburban gardens relative to agricultural and natural areas, because of the additional pollen and nectar availability provided by gardens (Kaluza et al., 2016). This increase in pollen and nectar could also be due to the prolonged duration of flowering in these gardens, which can span a longer time period relative to more natural flowering plant communities (Harrison & Winfree, 2015). On the other hand, many ornamentals do not provide enough pollen or nectar to sustain bee populations (Corbet et al., 2001; Garbuzov et al., 2017). Urban flower populations may not provide sufficient resources to attract and support large populations of wild bees unless they are planted with the specific needs of these bee species in mind (Lowenstein et al., 2019; Nicholls & Ibarra, 2017). Increased awareness of pollinator-friendly plants and designing

gardens to specifically support bees could help contribute to successful pollinator persistence in urban areas.

Between 1971-2011, Canada experienced a 157% increase in impervious surface land cover (Statistics Canada, 2016); this is expected to rise even further with population growth, causing continued losses of natural areas and untold impacts on wildlife, including wild bee communities (Deguines et al., 2012). Urbanization can alter the availability of bee foraging and nesting resources (Banaszak-Cibicka & Żmihorski, 2012), leading to changes in the community, including loss of individuals and species (Williams et al., 2010; Wilson & Jamieson, 2019). Previous studies of bees in urban areas have reported a wide variety of patterns. For example, a study of wild bees in the New York area found that the bee community in suburban areas was representative of the bee assemblage reported by a previous study at a nearby nature reserve (Fetridge et al., 2008). Another study of replicated bumblebee colonies in Surrey, UK, found that colonies in urban areas had higher reproductive success than those in the surrounding agricultural areas (Samuelson et al., 2018). However, these results contradict other studies of bees in urban areas, which have reported that the abundance of wild bees (Fortel et al., 2014) and the diversity of bumblebees (Ahrné et al., 2009) decreased with increasing impervious surface area in the region. These conflicting studies vary in important ways, such as the variables measured and the number of bee taxonomic groups included in the study, which could at least partially explain why they differ in terms of their outcomes. In this study, I focused on two key landscape variables, and assessed local site characteristics across an urban gradient, in order to determine how these variables drive bee community metrics.

While trees can provide valuable resources for pollinators, shading from trees is also known to alter bee foraging activity (McKinney & Goodell, 2010). Studies have shown declining bee abundance as canopy cover increases in both forest (Grundel et al., 2010) and agricultural environments (Hall et al., 2019) and these patterns also appear to hold in urban areas (Brunbjerg et al., 2018; Burdine & McCluney, 2019). The likely mechanism is deciduous tree leaf-out, which blocks light flow to the herbaceous layer below, reducing overall floral resources in forests (Hanula et al., 2015; Korpela et al., 2015). This shade also reduces air temperature, which is associated with fewer bees and lower activity levels (Korpela et al., 2015; Stone, 1994). Thus, although forests and heavily shaded areas are an important ecological and sociological component of urban planning, they may provide habitat for fewer bee species, relative to open meadows and gardens (Winfree et al., 2007).

The conservation status of most wild bee species is poorly understood (Cane & Tepedino, 2001), although many populations appear to be in decline, largely due to human activities (Cameron et al., 2011; Cariveau & Winfree, 2015). A lack of comprehensive insect surveys combined with broad local variation makes it difficult to define the conservation needs of bees in urban areas (Winfree et al., 2011). Despite this, landscape factors like impervious surface area and dense tree cover are likely to be negatively correlated with bee abundance and diversity.

The effects of urbanization on wild bee species are just starting to be investigated, with a mixture of results (Ahrné et al., 2009; Fetridge et al., 2008; Fortel et al., 2014; Samuelson et al., 2018). Contrarily, the impact of tree cover on bees in urban areas is poorly studied. Establishing the landscape features associated with wild bees in urban settings is an important

part of establishing conservation plans for these species. In my thesis, using surveys of wild bee communities in residential yards, I addressed whether local bee abundance, richness, and diversity vary according to urbanization (measured as percent impervious surface cover) and tree cover (measured as percent tree canopy cover) across an urban landscape.

Materials & Methods

Site Description & Selection

To gather a sufficient number of candidate yards to survey, I used social media advertising and flyer distribution to promote my study across the City of Ottawa. Through this polling, 380 volunteers applied to have their yards considered. To be considered for the study, participating yards had to include at least one tree, a minimum of 50% green space (grass or garden) in the candidate yard, zero use of pesticides and the yard had to be independently accessible from the house. To evaluate the criteria, volunteers filled out a survey (Figure A1) and the answers (presence of a tree, cardinal yard direction, and amount of green space) were corroborated with Google Maps satellite data (https://www.google.com/maps). Yards were required to be south or west facing to control for daily sun exposure, which is known to impact bee visitation rates (Korpela et al., 2015; Stone, 1994). Of the original volunteer sites, 120 met the control criteria.

Landcover was established using geographic information systems (GIS) data on impervious surface and tree canopy area, using ArcMap version 10.5 (Esri, 2019). Both datasets were obtained from the City of Ottawa and included vector data in the form of polygons. The impervious surface dataset was originally developed by Natural Resources Canada in 2013 and maps impervious surfaces throughout the city (i.e., sidewalks, roads, parking lots, buildings), at

a resolution of 20 cm. The tree canopy dataset (2019) was created through a collaboration between the National Capital Commission (NCC), Ville de Gatineau, and the City of Ottawa. This dataset shows the canopy of all shrubs and trees over one m tall using aerial imagery, light detection and ranging (LiDAR) data to a resolution of 20 cm. Candidate sites were then placed into four categories based on their land cover characteristics (method described below). The categories were: high urbanization, high tree cover (HUHT); low urbanization, low tree cover (LULT); high urbanization, low tree cover (HULT) and low urbanization, high tree cover (LUHT) (Figure 2.1). Using a spatial statistics tool in ArcGIS (Esri, 2019), I found the percent of urbanization and tree cover surrounding each candidate site at a 500 m radius (Figure 2.2). Final study sites were then selected from the list of candidate sites by choosing the sites that represented a wide range of the two landscape variables (for a detailed description of this method, see Flick et al., 2012). This was done to emphasize a gradient of urbanization and tree cover across the city. Unfortunately, I did not manage to break the correlation between these two landscape variables (Tables 2.5-2.6 & Figures 2.3-2.4). This may be problematic as it then difficult to separate which landscape factor explains which proportion of the variation (Graham, 2003). Despite this, it can be justifiable to include correlated variables in ecology, especially if they represent separate ecological processes as they can help explain the phenomenon in question (Smith et al., 2009) (Figure 2.5). Sites with 'high' urbanization (at 500 m) had a range of impervious surface cover between 36.2 – 69.8%, while sites in the 'low' urbanization category had a range of 2.9 – 31.8%. Sites with 'high' tree canopy cover sites ranged from 30.0 – 60.1% canopy cover, while those with 'low' tree cover ranged from 15.6 – 26.9%. Sites were then selected to be a minimum of one kilometer apart to minimize non-independence of bee

communities, since the majority of bees' range within a one-kilometre distance or less, from their nest (Greenleaf et al., 2007; Zurbuchen et al., 2010). Overall, a total of 27 sites were selected (Figure 2.6), with six in the HUHT, seven in the HULT, eight in the LUHT, and six in the LULT categories (see Tables B1 & B2 for a breakdown of landscape gradients surrounding each site at different scales). Plants at the selected sites consisted of exotic and ornamental plantings as well as combinations of city planted, and owner planted tree species (Tables C1 & D1). All sites were detached homes meant for single-family units with differing management styles and gardening activity levels. Most sites were dominated by expanses of green lawns, with gardens bordering the edges of the house; however, a few sites had converted their lawns to planted garden beds with little to no grass.

Bee Collection

Sites were visited to collect bee abundance and diversity data once a month from May – August 2019. Sampling occurred between the hours of 10:00-16:30 during days when it was sunny or brightly overcast and the wind speed did not surpass three m/s (Martins et al., 2018). Temperature and average wind speed measurements were taken at the end of the bee collection period, using a Kestrel 2000 Wind Meter (Nielsen-Kellerman, Boothwyn, PA, USA). The Kestrel was held approximately one m above the ground, at shoulder height, for one minute. To organize site surveys, two survey periods (morning and afternoon) were established. The morning period lasted from 10:00-12:00 and the afternoon period from 12:45-16:30. Site visits were alternatively rotated each month so that each site was sampled twice during each period.

Bees were captured through both pan trap and hand netting methods. Pan traps are an efficient bee sampling technique across taxa as they passively sample throughout the day and can reduce collector bias (Westphal et al., 2008). Pan traps were placed in two sets of three (hereafter, 'triplet') per site in a sunny area. Due to the differences in yard size and shape, it was not possible to create a standardized transect design for the pan traps. Triplets each contained a fluorescent white, blue and yellow bowl, filled to the brim with water and a drop of dish soap (Dawn dishwashing detergent), to break the surface tension. Triplets were placed at least 3 m apart from each other; bowls within the triplet were placed 20 cm apart. Pan traps were placed in areas that received full sun throughout the day and were near available floral resources (Lerman & Milam, 2016). Pan traps were left out at each site for 24-48 hours, after which the contents were placed in 95% ethanol and stored in a lab freezer (-6 ^{o}C) until they were pinned.

To make sure an even survey of the bee community was sampled, bees were also collected using a sweep net or vials in the surrounding floral resources for 30 minutes at each site visit. To reduce collector bias, the same collector sampled during every site visit. When sampling, the collector moved around the yard and captured bees when they were sighted. When a bee was caught, it was placed in ethanol without stopping the timer.

If a honeybee and a wild bee occurred on the same flower, preference was given to catch the wild bee first. To reduce the impact of sampling on the production of bumblebee colonies, bumblebee queens caught during May and June were put in a vial, photographed, and then released. These bumblebees were then identified based on the images captured.

Bee Identification

Collected bees were washed, blow-dried and pinned according to methods outlined in "The Very Handy Bee Manual" (Droege, 2015). When possible, bees were identified to species level using a variety of taxonomic keys and online resources (Table E1) (Ascher and Pickering 2015; Bouseman & LaBerge, 1978; Buck et al., 2005; Cooper, 1948; Droege et al., 2010; Gibbs, 2010; Gibbs et al.,2013; LaBerge, 1973; Laberge, 1980; LaBerge, 1985; LaBerge, 1987; LaBerge, 1989; LaBerge & Ribble, 1972; LaBerge & Ribble , 1975; Mcginley, 2003; Michener, 2000; Mitchell, 1962; Onuferko, 2018; Rehan & Sheffield, 2011; Rightmyer et al., 2010; Rightmyer et al., 2013; Romankova, 2003a, 2003b; Rowe, 2017.; Sheffield et al., 2011; Williams et al., 2014). Due to the difficulties in identifying bees to species in the genera *Lasioglossum* and *Sphecodes*, these bees were identified to morphospecies (Delphia et al., 2019). To prevent the overestimation of species counts, morphospecies were identified based on female specimens, except for in the case of *Lasioglossum* subgenera *Evylaeus* and *Hemihalictus*. In these subgenera, only the male specimens were identified to morphospecies. This was due to the similar characteristics in the male specimens that made them easier to attribute to a single species when compared to the female specimens in the same subgenus.

Since *Hylaeus modestus* and *H. affinis* are difficult to differentiate using morphological keys, they were considered to be a single species aggregate (Grundel et al., 2011). A single specimen of *Hylaeus* was found to have characteristics of both *H. modestus* and *H. annulatus*. This specimen was labelled *Hylaeus undetermined* and was not included in further analyses. Identification of *Lasioglossum* subgenera follows Gibbs et al. (2013). All specimens are housed at the University of British Columbia, Sargent lab collections. Bees that were only identified to the subgenus level (95 bees, 9% of total bees caught) were excluded from all analyses.

Bee abundance was recorded as the total number of identified bee species/morphospecies and bee richness was recorded as the total number of collected species (across all sampling events). Bee diversity was calculated using the Shannon's Diversity Index and was then transformed into effective numbers (Hill numbers) for ease of interpretation (Chao et al., 2014). Diversity metrics were calculated using the 'vegan' package (Oksanen, 2013). Bee ecological characteristics were calculated, but were not included as part of the analysis (see Appendix E).

Local Factors

All non-graminoid floral resources in the yard were counted during each site visit. Flowering heads (e.g., *Taraxacum*, *Trifolium*) were counted as a single floral unit. Floral units on other forms of inflorescence were counted according to Guezen et al. (2017) (i.e., five inflorescences were randomly selected and the flowers on these were tallied and then averaged). This average was then multiplied by the number of inflorescences to get an estimated total number of floral units for the plant. To obtain a sitewide measure of pollen and nectar availability, floral abundance was counted as the total number of floral units that appeared at each site throughout the whole season. Similarly, to obtain a measure of how variable pollen and nectar resources were, floral richness was counted as the total number of species occurring at a site across visits.

Plant species that couldn't be identified on site were photographed and identified at a later time. Wildflower species were identified by Newcomb (1977) and GoBotany Native Plant Trust (https://gobotany.nativeplanttrust.org/). Ornamental species were identified using garden catalogues (Farmer's Almanac, 2020; Syngenta Flowers, 2020). Plants were identified to

species when possible. Plants that were only identified to genus, were then sorted into morphospecies based on differing morphological characteristics.

Site area was measured using Google Maps (https://www.google.com/maps) and was included in the statistical analysis to account for the variation in size between sites. Nesting characteristics were measured at each site; however, they were not included in the analysis as site size was used instead as a more reliable proxy for nesting habitat (see Appendix F).

Statistical Analysis

First, the relevant scale of each landscape variable was chosen based on the methods described in Flick et al. (2012). Briefly, simple linear regressions were generated to show the relationship between each of the response variables (bee richness, bee abundance, and bee diversity) and each of the landscape variables (urbanization and tree cover), measured at each of the six spatial scales (Tables B1 & B2). The spatial scale which explained the most variation in bee richness and abundance was then determined by plotting the R^2 values against the landscape scale for each response gradient. Although the R^2 was low for all of the regressions, urbanization at 50 m and tree cover at 300 m exhibited the highest R^2 for bee richness and abundance, while bee diversity exhibited the highest R^2 at the urbanization level of 200 m and tree cover level of 100 m (Figure 2.7).

A multi-model selection process (Symonds & Moussalli, 2010) was then performed to determine the model that best explained variation in each of the two response variables. For bee richness and abundance, models containing combinations of the following predictor variables were tested: urbanization at 50 m, tree cover at 300 m, the log of floral abundance, floral richness, and site size. For bee diversity, the predictor variables examined were

urbanization at 200 m, tree cover at 100 m, the log of floral abundance, floral richness, and site size. All 31 combinations of multiple regressions were run for all three response variables and were then ranked by Akaike Information Criteria (AIC) values (Tables 2.2-2.4) (Burnham & Anderson, 2002).

Model residuals were visually assessed for the assumptions of linear regressions and showed no significant deviations from normality or heteroscedasticity. The exception was floral abundance, where a logarithmic transformation was applied to meet the condition of normality (Benoit, 2011). The models were checked for spatial autocorrelation by performing a Moran's I test on the residuals of the top five models for each response variable, using the 'spdep' package (Bivand & Wong, 2018). Spatial autocorrelation of the residuals was not detected ($P > 0.05$ for all models). The models were also tested for multicollinearity using a test for variance inflation factors (VIF) (Craney & Surles, 2002). The VIFs of the predictor variables of the top models were checked and recorded for each response variable (Tables 2.7-2.9). All VIFs were below five, indicating low multicollinearity among variables (Craney & Surles, 2002). This analysis was done using the 'car' package (Fox et al., 2012). All analyses were performed in R version 3.5.1 (R Core Team, 2020).

Results

Bee Community Composition

In total, I collected 1052 bees from five different families, for a total of 98 species and morphospecies (Table E1). Interestingly, no one species dominated the samples. The most abundant bees were *Halictus confusus* (Halictidae), *Ceratina calcarata* (Apidae) and the exotic *Chelostoma campanularum* (Megachilidae). Combined, these three species accounted for 17%

of all the bees caught. Approximately 20% of the collected bee species were represented by fewer than ten individuals. The average bee abundance at a site was 35.5 ± 4.4 (SE) individuals, average bee richness was 17.9 ± 1.44 (SE) species per site, and average bee diversity (as Hill numbers) was 14.0 ± 1.1 (SE) species per site.

Local Factors

I identified 409 animal-pollinated flowering plant species/morphospecies across my study sites, for an average of 45 +/- 3.6 (SE) flowering species per site across the season. For floral units, the overall average was 151,412 +/- 43,406 SE per site across all sampling days. The top three most common contributors to floral resources were sugar maples (*Acer saccharum.*), currant (*Ribes spp.*), and linden trees (*Tilia spp.*), which accounted for 40% of the total floral units (Table C1). The most common flowering plants across all sites were the common lawn weeds black medic (*Medicago lupulina),* dandelion *(Taraxacum spp.),* and white clover (*Trifolium repens).*

Multiple Regression Analyses

Following Flick et al., (2012), a total of 31 models were run on each response variable of bee abundance, richness, and diversity (Hill numbers) to determine the variables best explaining site to site variation (Tables 2.2-2.4). For bee abundance, the best model included urbanization, floral richness, and site size. The top three models had AIC scores within two units of one another. As predicted, urbanization was negatively correlated with bee abundance, while floral richness was positively associated with bee abundance, richness, and diversity. Bee abundance was negatively correlated with site size. The three top models for bee abundance have R^2 values ranging from 47-49% (Table 2.2). For bee richness, the best model only included floral richness as the predictor variable. The top five models are within two AIC units of the top

model, with floral richness having a positive effect in all models, as expected. The five top models have R^2 values ranging from 28-31% (Table 2.3). Consistent with bee species richness, the best model for bee diversity was floral richness as the only predictor variable. The top five models were within 2 AIC units from the top model. The R^2 values for these models ranged from 30-33% (Table 2.4). For all three response variables (bee abundance, richness and diversity), floral richness was a constant in all top models, with no other predictor showing as strong a response. Neither tree cover, nor floral abundance were consistently present in any of the top models.

Effect sizes were interpreted based on the unstandardized coefficients, with one unit equalling one bee (or one bee species or unit of diversity). For bee abundance, on average, a one unit increase in urbanization resulted in a decline of 0.5 in bee abundance, while a one unit increase in site floral richness was associated with an average increase of 0.7 in bee abundance. Site size was negatively associated with bee abundance, although with a very small average effect size (e.g., <0.06 for bee abundance). Floral richness was positively associated with an effect size of 0.2 on bee richness in the top model. Similar to bee richness, bee diversity (in Hill numbers) was explained by a model that included only floral richness with a positive effect of 0.17.

Discussion

As urbanization continues to consume existing wild, semi-wild, and agricultural lands, there is growing concern that wild bee species will be negatively impacted by declining habitat availability (Potts, 2010). In a study of wild bee community metric drivers in residential yards, spanning a gradient of urbanization and tree cover, I found that a yard's floral richness was the

most consistent correlate of bee abundance, richness, and diversity. While urbanization was negatively associated with bee abundance, neither the degree of urbanization nor tree cover appeared in the top models describing bee richness or diversity. My thesis supports a growing body of work showing that local factors are a key driver of bee populations across ecosystems. These findings suggest that conservation strategies need to target the floral composition of residential yards, to promote bee conservation in urban areas.

Bee Community Response to Landscape Factors

I hypothesized that increased urbanization (measured as impervious surface within a set area) would be associated with lower bee abundance, richness, and diversity, and my findings at least partially support this. Urbanization was negatively correlated with bee abundance in the top three models. In the top model, the effect size of urbanization showed a decline of 0.5 bees for every unit of increase in percent urbanization within 50 m. On the other hand, urbanization was not a top predictor of either bee species richness or diversity. There are several reasons why the abundance result did not mirror the findings for species richness or diversity. For example, impacts of habitat loss on species diversity may lag its impacts on abundance through factors such as extinction debt (Kuussaari et al., 2009). Areas with higher habitat homogenization may host a relatively large number of individual bees, but with fewer species. Following this reasoning, it could be that the negative relationship between bee abundance and urbanization detected here, is an early warning of future species loss.

Tree cover did not appear in any of the top models. These results suggest that tree cover is not an important driver of bee distributions in our survey area. Locally, canopy cover has been shown to be correlated with low bee abundance in forest (Grundel et al., 2010) and

urban environments (Burdine & McCluney, 2019). The explanations offered for this pattern are that the shade from tree cover reduces floral resource availability (Matteson et al., 2013) and may also impede a bee's thermoregulatory abilities (Polatto et al., 2014). Yet, many flowering trees (e.g., apple, cherry, and maple) can provide more abundant floral resources in a smaller area than other garden plants (Donkersley, 2019) and can be a rich source of pollen and nectar in neighbourhoods dominated by expanses of lawn (Somme et al., 2016). For bee species that emerge in early spring (e.g., bees in the genus *Andrena*), before most other resources are in bloom, flowering trees are essential for colony initiation (Moquet et al., 2015) and survival (Donkersley, 2019; Hausmann et al., 2016).

On a landscape scale, tree cover may not have figured prominently as a predictor due to conflicting effects of tree presence on bee diversity measures, or because bees may respond more strongly to local (e.g., floral richness) rather than landscape factors (e.g., tree cover). For example, some studies have found that landscape variables are not correlated with increases in bee abundance/ richness when compared to local variables in residential gardens (Quistberg et al., 2016). While studies looking at tree cover at the landscape level in urban environments are lacking, previous studies in forest/scrub and savanna/woodland habitats in Indiana (Grundel et al., 2010), as well as in forest/meadow habitats in northern Arizona (McCabe et al., 2019), found that local bee abundance declined as tree cover increased. It is likely that the impacts of tree cover on bee abundance, richness, and diversity, most often studied in forests and other natural ecosystems, differs in urban landscapes. This may be because urban trees largely depend on human selection, resulting in different tree species compositions when compared to natural environments, changing the resource availability for bees. Tree canopy cover in urban

areas also tends to be lower overall, relative to forest; the impacts of shading may not factor in at the moderate levels of canopy cover (average 29 %) that occurred at my sites.

Bee Community Response to Local Factors

In this study, floral richness was the most consistent predictor of all three bee community metrics. Floral abundance (Bates et al., 2011), richness (Hülsmann et al., 2015), and floral richness (Ahrné et al., 2009; Bates et al., 2011) have previously been shown to be important predictors of bee abundance and diversity in urban environments. In my study, floral abundance was not consistently retained in the top models for bee abundance, richness, or diversity. While other studies have found no significant relationship between floral abundance/richness and bee abundance/richness (Choate et al., 2018; Geslin et al., 2016), others have found it to be an important predictor of bee abundance (Bates et al., 2011; Plascencia & Philpott, 2017). One possible explanation is the compositional make-up of the gardens. A few of our surveyed sites had many blooms from plants that had low bee attractiveness due to a decrease in pollen and/or nectar production (Table C1). This may cause an increase in floral abundance that would not correlate with bee abundance, richness, and diversity. It may also be why floral richness was a better indicator of bee diversity measures as yards with a greater variety of bee attractive species had higher bee abundances. Unfortunately, there can be a disconnect between the types of plants residential garden owners prefer and those that provide resources for insects, including bees (Garbuzov & Ratnieks, 2014). To prevent this, urban green spaces should be designed to contain a variety of different plant species, and not rely wholly on one or a few highly abundant floral species.

Scale of Effect

Analysing local and landscape variables across multiple scales are especially important in urban environments as land-cover can be highly variable over small spatial extents (Luck & Wu, 2002). In general, the landscape factors I chose (impervious surface and tree cover) explained very little of the overall variation in bee abundance or diversity. However, there was a distinct peak for each factor in terms of the proportion of total variance in bee metrics explained (Figure 2.7). For bee abundance and richness, the peak R^2 was at 50 m for urbanization and 300 m for tree cover, while for bee diversity the peak R^2 was at 200 m for urbanization and 100 m for tree cover (Figure 2.7). These findings were similar to those from a study in Birmingham, UK that found that bee richness was best explained by percent vegetation cover (both ground and tree cover) in a major urban city at 100 m (Brunbjerg et al., 2018). A San Diego, CA study looking at impervious surface area at multiple spatial scales to see the impact of bumblebee abundance, showed the strongest effects of impervious surface at 750 m for *B. californicus* and 250 m for *B. vosnesenskii* (Shochet et al., 2016). Although these studies do find peaks at different spatial scales, it does suggest that wild bees may respond to the environment on a scale more similar to their daily foraging range.

For most bee species, average and maximum foraging ranges are unknown, although some generalities have been suggested (Greenleaf et al., 2007; Zurbuchen et al., 2010). A British study by Knight et al. (2005) found that the maximum foraging range of different *Bombus* spp. ranged from 450 m – 758 m, despite their recorded maximum travelling range of ~5 km (Osborne et al., 2008). Despite a large maximum range, daily movements typically occur on a much smaller scale (Walther-Hellwig & Frankl, 2000). Bumblebees are a large eusocial species, but most bee species, including the main ones in my study, are solitary, small to

medium in size, and have limited dispersal ability (Greenleaf et al., 2007). This dictates the range at which they interact with floral resources (Ogilvie & Forrest, 2017). Therefore, it is likely that the daily range of wild bees is very small, possibly restricting them to a single residential yard. This indicates that improving local factors may be a more beneficial action for their preservation (Steckel et al., 2014; Torné-Noguera et al., 2014).

Limitations & Future Research

This was an observational study that took place over a single summer. While I tried to capture aspects of urban areas that most impact wild bees, other factors like weather patterns (Radmacher & Strohm, 2011), socioeconomic status of the neighbourhood (Luliano et al. 2017), and choice of plantings (Salisbury et al., 2015) could also affect bee community compositions in these areas. It is also likely that local bee communities respond differently to urbanization and tree cover.

Pan traps are an effective passive sampling technique used in pollination ecology. However, they have been shown to oversample certain species. Genera like *Halictus* and *Lasioglossum* are known to be oversampled by pan traps while genera like *Colletes* and *Andrena* tend to be under-sampled, probably because they are less attracted to the traps (Roulston et al., 2008). Bees that have larger body sizes (e.g., bumblebees) also have a reduced collection rate as they can escape (Cane et al., 2001; Roulston et al., 2008), potentially adding bias. Pan traps can also oversample sites with limited floral resources as they become highly attractive in a barren landscape (Cane et al., 2000). Since my sites differed in management and gardening styles, pan traps could have oversampled at sites with limited floral resources. In the present study, sweep netting was included to account for pan trap shortcomings.

The level of garden management undertaken at each site may influence the bee community composition. Yards with ruderal lawn species like dandelions and white clover may sustain larger bee populations as these flowers are a key resource for bees (Roulston 2000). In contrast, yards that are frequently mowed or weeded, prevent ruderal flower growth and may also discourage ground nesting bees from establishing nests due to these increased disturbances (Tonietto et al., 2011). Garden management is also influenced by socioeconomic status. Previously, household income has been positively correlated with yard floral abundance (Baldock et al., 2019). In addition, residents with a higher than average socioeconomic standing may be more likely to volunteer for studies such as this one, as they have the time and means to increase their garden's biodiversity. Although it would be more work, random identification of households in different land use categories followed by door knocking would help to alleviate the background effect of socioeconomic status on site selection.

Future research should continue to quantify to what extent bees use urban trees during their active season. Specifically, separating how bees differ in their use of resources from wind-pollinated versus insect-pollinated trees, is an avenue that has previously been underexplored. This information could help to provide tree species recommendations for city planners, which may increase the floral resource availability for bee communities.

Conclusion

Overall, I found that a site's floral richness was a better predictor of bee abundance, richness, and diversity than urbanization or tree cover. Urbanization did have a negative relationship with bee abundance, but not with bee species richness or diversity. Accordingly, tree cover did not factor in as a top predictor of any response variables. My results support other studies in

finding that high site floral richness may help to mitigate the negative impacts of urbanization (Ebeling et al., 2008; Hülsmann et al., 2015). Recent public awareness of issues surrounding pollinators has increased community engagement for projects aiding in bee conservation. My results support the idea that planting additional flowers in residential yards improves conditions for bee populations. These are small-scale changes occurring on a local scale that everyone can participate in. By promoting these ideas and designing cities to support wildlife, we can help prevent the loss of these wild bee species.

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Chapter 3: Conclusions

Declining bee populations are of global concern (Potts et al., 2010), due to their role as the primary pollinators of most flowering species (Ollerton et al., 2011). Human population growth is leading to an increasingly urbanized environment, removing natural habitat that is crucial for wild bee survival (Vanbergen, 2013). In fact, the City of Ottawa recently approved a large increase in the size of its urban boundary, which could drive a decline in bee abundance in areas under development (City of Ottawa, 2020). Urban systems are often solely dependent on wild bees for their pollination services (Ballare et al., 2019), which makes them an important resource to protect. In many cases, pollinator conservation can be encouraged by educating homeowners and city planners on bee attractive flora (Garbuzov et al., 2017).

I hypothesized that landscape gradients of urbanization and tree cover would be associated with lower bee richness and abundance, due to the loss of habitat associated with the former, and the negative impact of the latter on flowering plants. Surprisingly, these landscape gradients were not strongly associated with among site variation in bee communities. The local factor of floral richness was a much stronger predictor of bee abundance, richness, and diversity than either of the landscape gradients. This is an exciting finding as it suggests that homeowners diversifying floral species planted in their yards can help to maintain wild bee communities in urban areas. These results are supported by previous research that has found that cities are a pollinator refuge (McFrederick & LeBuhn, 2006; Threlfall et al., 2015); however, this is only possible if public communities are engaged and encouraged to maximize biodiversity on their private property.

The conclusions of this study highlight many gaps that could be avenues for future research. Such gaps include investigating how shade trees impact bees in urban environments and whether supplemental plantings of a diversity of flowering plant species is associated with increased bee community abundance and diversity. Commonly sold garden plants should also be surveyed for their ability to provide nectar and pollen resources as this would help identify the flowers homeowners should be encouraged to plant. Garden centres could also be encouraged to stock and advertise bee-friendly plantings. Overall, a major data gap is the lack of comprehensive species lists for many urban areas. It is difficult to separate which species are declining due to human activity when we do not know what species originally occurred in these areas. Studies focusing on quantifying a baseline for taxonomic populations in rural areas are needed to provide a comparison to species living in urban ones.

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Tables & Figures

Table 2.1. Sizes of the surveyed residential yards

Site	Site Size (m^2)
S1	201.44
S2	133.43
S3	182.8
S4	95.76
S5	92.31
S6	279.69
S7	265.7
S8	110.29
S9	290.29
S10	102.34
S11	321.74
S12	70.03
S13	118.27
S14	406.97
S15	86.79
S16	539.45
S17	116.27
S18	190.72
S19	120.81
S20	228.25
S21	133.59
S22	752.7
S23	193.52
S24	185.53
S25	124.48
S26	442.81
S27	204.54

Table 2.2. All models comparing bee abundance against predictor variables. Models are in order of the lowest AIC values. Hyphens indicate that the variable was absent in the model. Values in brackets indicate standardized coefficients (Z scores)

Model Number	AIC[1]	R^2	df	P-value	Impervious Surface (50 m)	Tree Cover (300 m)	log (Floral Abundance)	Floral Richness	Site Size
M2.2.1	224.06	0.466	23	0.00209	-0.511 (-0.389)	-	-	0.774 (0.797)	-0.0566 (-0.504)
M2.2.2	224.94	0.487	22	0.00409	-0.493 (-0.375)	-	1.96 (0.154)	0.741 (0.763)	-0.058 (-0.516)
M2.2.3	226.02	0.466	22	0.00612	-0.524 (0.399)	-0.0532 (-0.0330)	-	0.777 (0.801)	-0.056 (-0.498)
M2.2.4	226.32	0.326	25	0.00187	-	-	-	0.554 (0.571)	-
M2.2.5	226.43	0.372	24	0.00378	-	-	-	0.654 (0.674)	-0.0267 (-0.237)
M2.2.6	226.89	0.488	21	0.0104	-0.477 (-0.363)	0.0574 (0.0350)	2.12 (0.167)	0.734 (0.756)	-0.0589 (-0.524)
M2.2.7	227.15	0.401	23	0.00732	-	-	2.25 (0.177)	0.621 (0.640)	-0.0295 (-0.262)
M2.2.8	227.52	0.346	24	0.00614	-	-	1.85 (0.145)	0.519 (0.534)	-
M2.2.9	227.84	0.338	24	0.00708	-0.143 (-0.109)	-	-	0.556 (0.573)	-
M2.2.10	228.29	0.327	24	0.00865	-	-0.0476 (-0.0290)	-	0.560 (0.577)	-
M2.2.11	228.3	0.375	23	0.0116	-	0.0962 (0.0590)	-	0.653 (0.673)	-0.0293 (-0.261)
M2.2.12	228.51	0.415	22	0.0154	-	0.226 (0.138)	2.82 (0.222)	0.610 (0.628)	-0.0364 (-0.324)
M2.2.13	229.18	0.354	23	0.0165	-0.12 (-0.0910)	-	1.68 (0.132)	0.524 (0.539)	-
M2.2.14	229.52	0.346	23	0.0189	-	0.0154 (0.00090)	1.88 (0.148)	0.516 (0.532)	-
M2.2.15	229.56	0.345	23	0.0192	-0.193 (-0.147)	-0.151 (-0.0920)	-	0.575 (0.592)	-
M2.2.16	231.12	0.356	22	0.0391	-0.148 (-0.113)	-0.0772 (-0.0470)	1.48 (0.116)	0.537 (0.553)	-
M2.2.17	234.77	0.0785	25	0.157	-	-	3.56 (0.280)	-	-
M2.2.18	236.16	0.0991	24	0.286	-	0.24 (0.147)	3.94 (0.310)	-	-
M2.2.19	236.66	0.0824	24	0.357	-0.0827 (-0.0630)	-	3.46 (0.272)	-	-
M2.2.20	236.72	0.00966	25	0.626	-0.129 (-0.0980)	-	-	-	-
M2.2.21	236.77	0.0785	24	0.375	-	-	3.59 (0.282)	-	-0.000897 (-0.00800)
M2.2.22	236.79	0.00713	25	0.675	-	0.138 (0.0840)	-	-	-
M2.2.23	236.9	0.00306	25	0.784	-	-	-	-	0.00622 (0.0550)

Model Number	AIC[1]	R^2	df	P-value	Impervious Surface (50 m)	Tree Cover (300 m)	log (Floral Abundance)	Floral Richness	Site Size
M2.2.24	237.93	0.107	23	0.449	-	0.318 (0.195)	4.35 (0.342)	-	-0.0114 (-0.101)
M2.2.25	238.16	0.0991	23	0.484	0.00318 (0.0020)	0.242 (0.148)	3.94 (0.310)	-	-
M2.2.26	238.59	0.0847	23	0.556	-0.124 (-0.0940)	-	3.58 (0.281)	-	-0.00668 (-0.0590)
M2.2.27	238.65	0.012	24	0.865	-0.101 (-0.0770)	0.0868 (0.0530)	-	-	-
M2.2.28	238.72	0.00966	24	0.89	-0.128 (-0.0970)	-	-	-	0.00024 (0.00200)
M2.2.29	238.77	0.00764	24	0.912	-	0.121 (0.0740)	-	-	0.0028 (0.0250)
M2.2.30	239.89	0.108	22	0.622	-0.0597 (-0.0450)	0.3 (0.183)	4.3 (0.338)	-	-0.0136 (-0.121)
M2.2.31	240.65	0.0121	23	0.963	-0.108 (-0.0820)	0.0915 (0.0560)	-	-	-0.00143 (-0.0130)

[1] AIC values are based off of unstandardized data

Table 2.3. All models comparing bee richness against predictor variables. Models are in order of the lowest AIC value. Hyphens indicate that the variable was absent in the model. Values in brackets indicate standardized coefficients (Z scores)

Model Number	AIC[1]	R^2	df	P-value	Impervious Surface (50 m)	Tree Cover (300 m)	log (Floral Abundance)	Floral Richness	Site Size
M2.3.1	181.45	0.283	25	0.00430	-	-	-	0.218 (0.532)	-
M2.3.2	182.60	0.306	24	0.0126	-	-	0.835 (0.155)	0.202 (0.493)	-
M2.3.3	182.85	0.299	24	0.0141	-	-	-	0.243 (0.593)	-0.00665 (-0.140)
M2.3.4	183.29	0.287	24	0.0172	-0.0362 (-0.0650)	-	-	0.219 (0.533)	-
M2.3.5	183.45	0.283	24	0.0185	-	-0.00377 (-0.00500)	-	0.218 (0.533)	-
M2.3.6	183.55	0.332	23	0.0237	-0.128 (-0.231)	-	-	0.273 (0.666)	-0.0142 (-0.298)
M2.3.7	183.73	0.327	23	0.0255	-	-	0.943 (0.176)	0.229 (0.559)	-0.00783 (-0.165)
M2.3.8	184.51	0.308	23	0.0347	-0.0250 (-0.0450)	-	0.800 (0.149)	0.203 (0.495)	-
M2.3.9	184.54	0.307	23	0.0351	-	0.0261 (0.0380)	0.890 (0.166)	0.198 (0.483)	-
M2.3.10	184.55	0.356	22	0.0388	-0.120 (-0.216)	-	0.873 (0.163)	0.258 (0.630)	-0.0148 (-0.311)
M2.3.11	184.77	0.301	23	0.0384	-	0.0333 (0.0480)	-	0.242 (0.592)	-0.00757 (-1.59)
M2.3.12	185.24	0.289	23	0.0462	-0.0455 (-0.0820)	-0.0280 (-0.0410)	-	0.222 (0.542)	-
M2.3.13	185.26	0.339	22	0.0499	-	0.0866 (0.125)	1.16 (0.216)	0.225 (0.549)	-0.0105 (-0.221)
M2.3.14	185.54	0.332	22	0.0551	-0.129 (-0.232)	-0.00339 (-0.00500)	-	0.273 (0.666)	-0.0141 (-0.297)
M2.3.15	186.41	0.359	21	0.0759	-0.107 (-0.192)	0.0490 (0.0710)	1.00 (0.187)	0.253 (0.616)	-0.0155 (-0.326)
M2.3.16	186.50	0.308	22	0.0767	-0.0200 (-0.0360)	0.0136 (0.0200)	0.835 (0.156)	0.201 (0.490)	-
M2.3.17	188.23	0.0783	25	0.157	-	-	1.50 (0.280)	-	-
M2.3.18	189.48	0.104	24	0.269	-	0.112 (0.162)	1.68 (0.313)	-	-
M2.3.19	190.06	0.0137	25	0.560	-	-	-	-	0.00557 (0.117)
M2.3.20	190.14	0.0814	24	0.361	-	-	1.43 (0.267)	-	0.00272 (0.0570)
M2.3.21	190.16	0.00990	25	0.622	-	0.0687 (0.0990)	-	-	-
M2.3.22	190.22	0.0787	24	0.374	-0.0106 (-0.0190)	-	1.49 (0.277)	-	-
M2.3.23	190.35	0.00303	25	0.785	-0.0305 (-0.0550)	-	-	-	-
M2.3.24	191.38	0.107	23	0.447	0.0366 (0.0660)	0.133 (0.192)	1.76 (0.327)	-	-
M2.3.25	191.46	0.104	23	0.461	-	0.121 (0.175)	1.72 (0.321)	-	-0.00126 (-0.0260)

Model Number	AIC[1]	R^2	df	P-value	Impervious Surface (50 m)	Tree Cover (300 m)	log (Floral Abundance)	Floral Richness	Site Size
M2.3.26	191.97	0.0169	24	0.815	-	0.0427 (0.0620)	-	-	0.00437 (0.0920)
M2.3.27	192.06	0.0139	24	0.846	0.00722 (0.0130)	-	-	-	0.00591 (0.124)
M2.3.28	192.13	0.0816	23	0.573	0.00871 (0.0160)	-	1.43 (0.267)	-	0.00313 (0.066)
M2.3.29	192.16	0.0101	24	0.885	-0.00959 (-0.017)	0.0638 (0.0920)	-	-	-
M2.3.30	193.38	0.107	22	0.627	0.0371 (0.0670)	0.132 (0.192)	1.76 (0.327)	-	9.03E-05 (0.00200)
M2.3.31	193.96	0.0176	23	0.937	0.0173 (0.0310)	0.0474 (0.0690)	-	-	0.00504 (0.106)

[1] AIC values are based off of unstandardized data

Table 2.4. All models comparing bee diversity against predictor variables. Models are in order of the lowest AIC value. Hyphens indicate that the variable was absent in the model. Values in brackets indicate standardized coefficients (Z scores)

Model Number	AIC[1]	R^2	df	P-value	Impervious Surface (200 m)	Tree Cover (100 m)	log (Floral Abundance)	Floral Richness	Site Size
M2.4.1	166.96	0.301	25	0.00305	-	-	-	0.174 (0.549)	-
M2.4.2	167.96	0.327	24	0.00872	-	-	0.686 (0.165)	0.161 (0.507)	-
M2.4.3	168.89	0.303	24	0.0132	-	-0.0196 (-0.0450)	-	0.178 (0.560)	-
M2.4.4	168.90	0.303	24	0.0133	-	-	-	0.180 (0.568)	-0.00162 (-0.0440)
M2.4.5	168.95	0.301	24	0.0135	0.00777 (0.0190)	-	-	0.174 (0.547)	-
M2.4.6	169.77	0.331	23	0.0239	-	-0.0315 (-0.0720)	0.733 (0.176)	0.166 (0.523)	-
M2.4.7	169.81	0.330	23	0.0243	-	-	0.721 (0.173)	0.170 (0.535)	-0.00252 (-0.0680)
M2.4.8	169.95	0.327	23	0.0258	0.00530 (0.0130)	-	0.684 (0.165)	0.161 (0.506)	-
M2.4.9	170.87	0.303	23	0.0370	-	-0.0141 (-0.0320)	-	0.180 (0.569)	-0.000970 (-0.0260)
M2.4.10	170.89	0.303	23	0.0372	-0.00276 (-0.00700)	-0.0212 (-0.0490)	-	0.178 (0.562)	-
M2.4.11	170.90	0.303	23	0.0374	-0.00349 (-0.00900)	-	-	0.181 (0.571)	-0.00183 (-0.0500)
M2.4.12	171.73	0.332	22	0.0548	-0.0154 (-0.0380)	-0.0413 (-0.0950)	0.754 (0.181)	0.168 (0.530)	-
M2.4.13	171.73	0.332	22	0.0549	-	-0.0233 (-0.0540)	0.741 (0.178)	0.170 (0.535)	-0.00147 (-0.0400)
M2.4.14	171.76	0.331	22	0.0555	-0.0171 (-0.0420)	-	0.743 (0.179)	0.174 (0.549)	-0.00361 (-0.0980)
M2.4.15	172.86	0.304	22	0.0811	-0.00955 (-0.0240)	-0.0175 (-0.0400)	-	0.183 (0.577)	-0.00141 (-0.0380)
M2.4.16	173.61	0.335	21	0.104	-0.0299 (-0.0740)	-0.0346 (-0.0790)	0.789 (0.190)	0.178 (0.560)	-0.00287 (-0.0780)
M2.4.17	174.20	0.0859	25	0.138	-	-	1.22 (0.293)	-	-
M2.4.18	175.50	0.0410	25	0.311	-	-	-	-	0.00745 (0.203)
M2.4.19	175.62	0.106	24	0.262	-	-	1.08 (0.261)	-	0.00530 (0.144)
M2.4.20	176.15	0.0876	24	0.333	-	0.0186 (0.0430)	1.18 (0.284)	-	-
M2.4.21	176.16	0.0875	24	0.334	0.0161 (0.0400)	-	1.21 (0.291)	-	-

Model Number	AIC[1]	R^2	df	P-value	Impervious Surface (200 m)	Tree Cover (100 m)	log (Floral Abundance)	Floral Richness	Site Size
M2.4.22	176.34	0.0105	25	0.611	-	0.0447 (0.103)	-	-	-
M2.4.23	176.46	0.0773	24	0.381	0.0908 (0.224)	-	-	-	0.0118 (0.321)
M2.4.24	176.55	0.00306	25	0.784	0.0224 (0.0550)	-	-	-	-
M2.4.25	177.01	0.125	23	0.370	0.0686 (0.170)	-	0.957 (0.230)	-	0.00883 (0.240)
M2.4.26	177.49	0.0411	24	0.604	-	-0.00625 (-0.0140)	-	-	0.00774 (0.210)
M2.4.27	177.57	0.107	23	0.447	-	-0.0207 (-0.0470)	1.10 (0.265)	-	0.00623 (0.169)
M2.4.28	177.91	0.0264	24	0.726	0.0594 (0.147)	0.0776 (0.178)	-	-	-
M2.4.29	177.99	0.0931	23	0.514	0.0354 (0.0880)	0.0396 (0.0910)	1.12 (0.269)	-	-
M2.4.30	178.38	0.0798	23	0.582	0.0987 (0.244)	0.0277 (0.0640)	-	-	0.0107 (0.296)
M2.4.31	179.01	0.126	22	0.545	0.0704 (0.174)	0.00557 (0.0130)	0.949 (0.228)	-	0.00867 (0.236)

[1] AIC values are based off of unstandardized data

Table 2.5. Pairwise correlations for predictor variables used in bee abundance and richness models

	Site Size (m^2)	Floral Richness	Floral Abundance	Urban 50m
Site Size (m^2)				
Floral Richness	0.43			
Floral Abundance	0.16	0.060		
Urban 50m	-0.55	0.017	-0.014	
Tree cover 300m	0.41	0.20	-0.12	-0.41

Table 2.6. Pairwise correlations for predictor variables used in bee diversity models

	Site Size (m^2)	Floral Richness	Floral Abundance	Urban 200m
Site Size (m^2)				
Floral Richness	0.43			
Floral Abundance	0.16	0.060		
Urban 200m	-0.53	0.061	0.043	
Tree cover 100m	0.56	0.26	0.18	-0.51

Table 2.7. VIF for models with R^2 values within 10% of the top model for bee abundance

Model Number	Impervious Surface (200m)	Tree (100m)	Log (Floral Abundance)	Floral Richness	Site Size
M2.2.1	1.61	-	-	1.39	1.99
M2.2.2	1.62	-	1.09	1.45	2.00
M2.2.3	1.74	1.29	-	1.41	2.04
M2.2.6	1.80	1.50	1.27	1.50	2.07

Table 2.8. VIF for models with R^2 values within 10% of the top model for bee richness

Model Number	Impervious Surface (200m)	Tree (100m)	Log (Floral Abundance)	Floral Richness	Site Size
M2.3.2	-	-	1.07	1.07	-
M2.3.3	-	-	-	1.23	1.23
M2.3.4	1.00	-	-	1.00	-
M2.3.5	-	1.04	-	1.04	-
M2.3.8	1.02	-	1.09	1.07	-
M2.3.9	-	1.12	1.15	1.15	-
M2.3.11	-	1.20	-	1.23	1.42
M2.3.12	1.22	1.26	-	1.05	-
M2.3.16	1.32	1.45	1.25	1.19	-

Table 2.9. VIF for models with R^2 values within 10% of the top model for bee diversity

Model Number	Impervious Surface (200m)	Tree (100m)	Log (Floral Abundance)	Floral Richness	Site Size
M2.4.2	-	-	1.07	1.07	-
M2.4.3	-	1.07	-	1.07	-
M2.4.4	-	-	-	1.23	1.23
M2.4.5	1.00	-	-	1.00	-
M2.4.7	-	-	1.09	1.27	1.25
M2.4.8	1.01	-	1.07	1.07	-
M2.4.9	-	1.45	-	1.23	1.66
M2.4.10	1.44	1.55	-	1.14	-
M2.4.11	1.62	-	-	1.45	1.99
M2.4.15	1.85	1.65	-	1.49	2.13

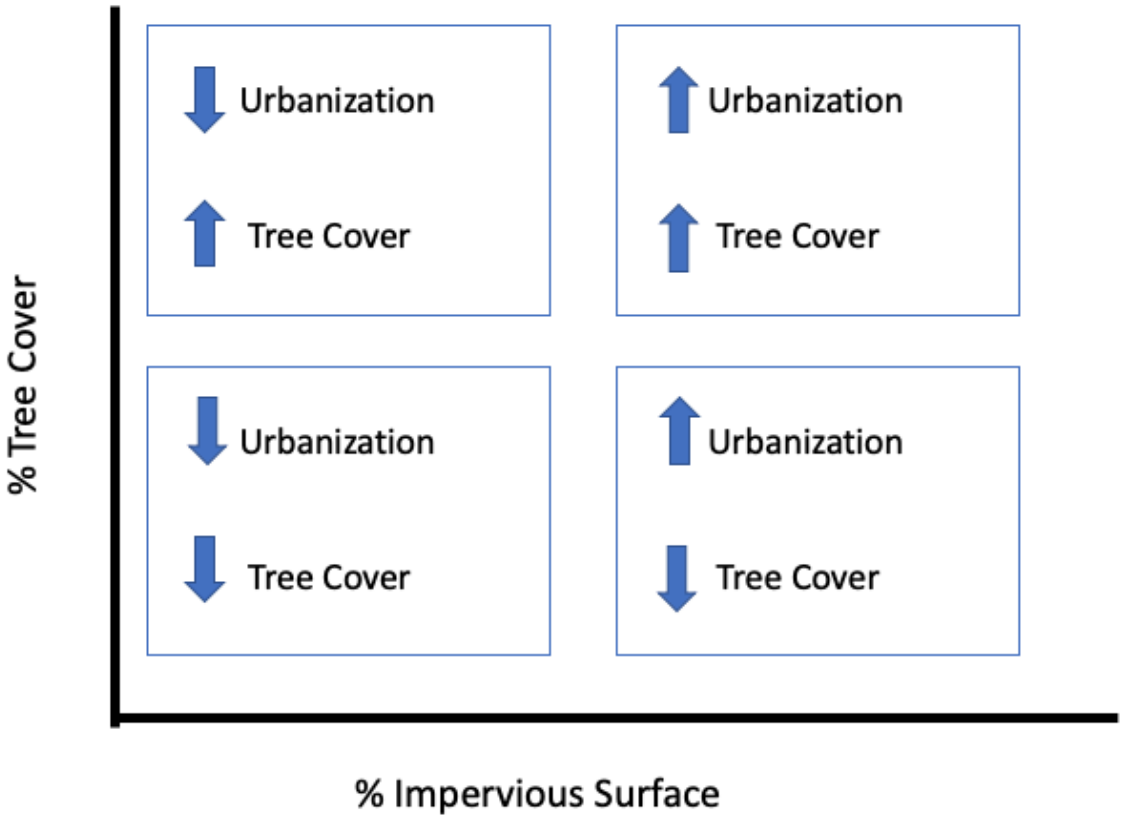

Figure 2.1. The categories used for site selection, based on the two landscape gradients, percent impervious surface area and tree cover

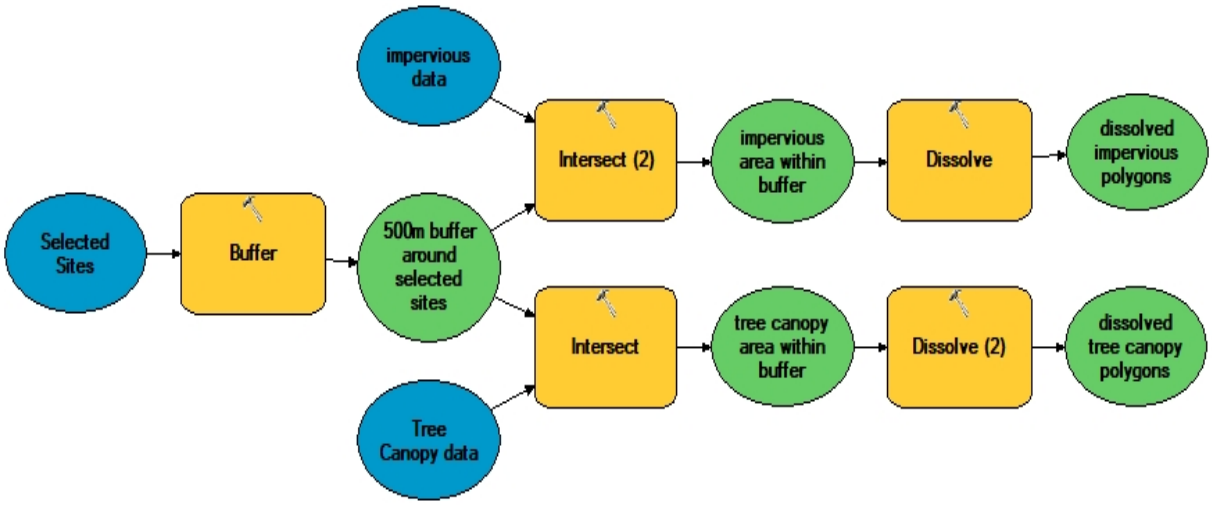

Figure 2.2. The model used in ArcGIS to obtain the levels of urbanization and tree cover at each site

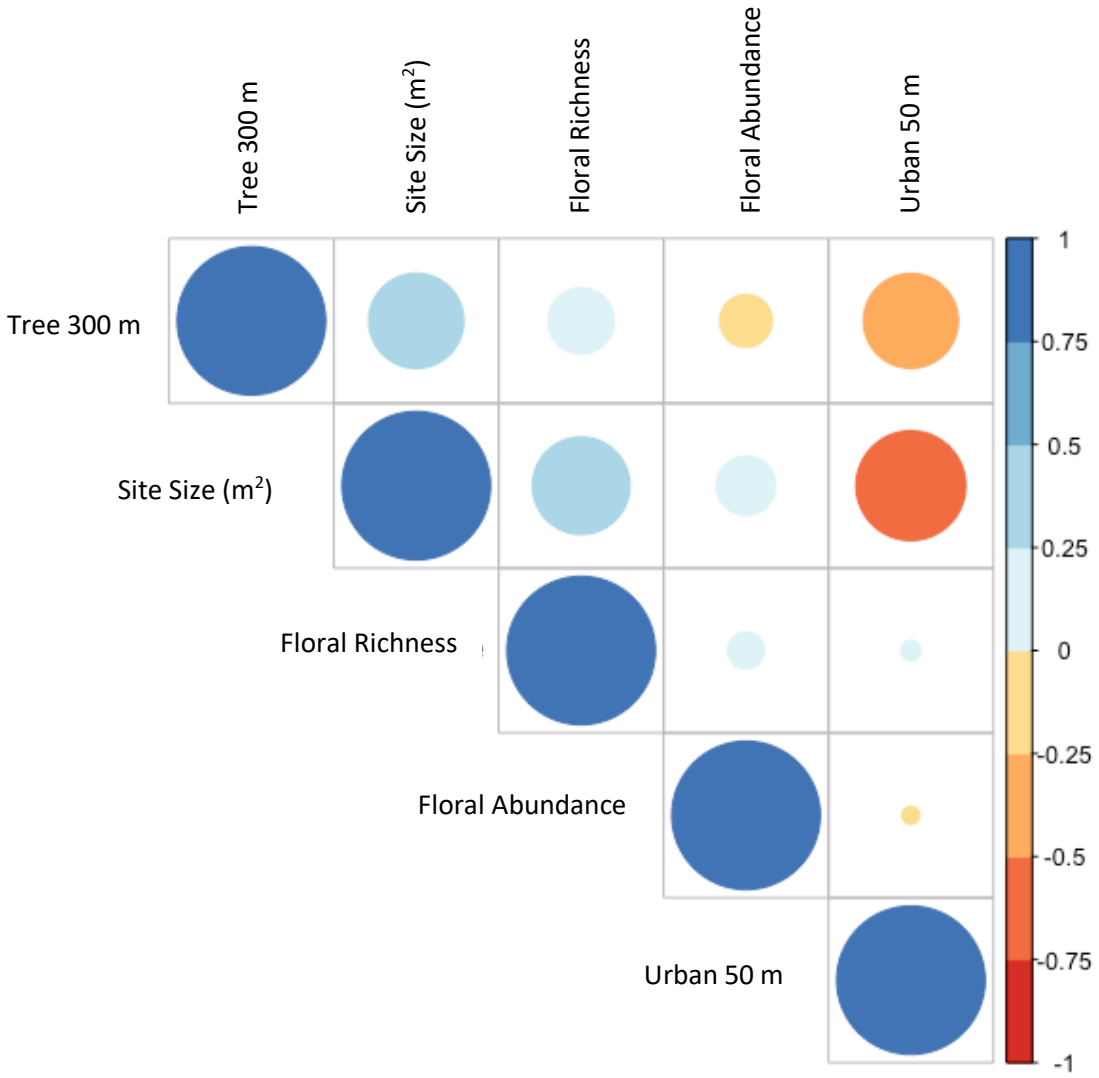

Figure 2.3. Correlation matrix for predictor variables used in bee abundance and richness models

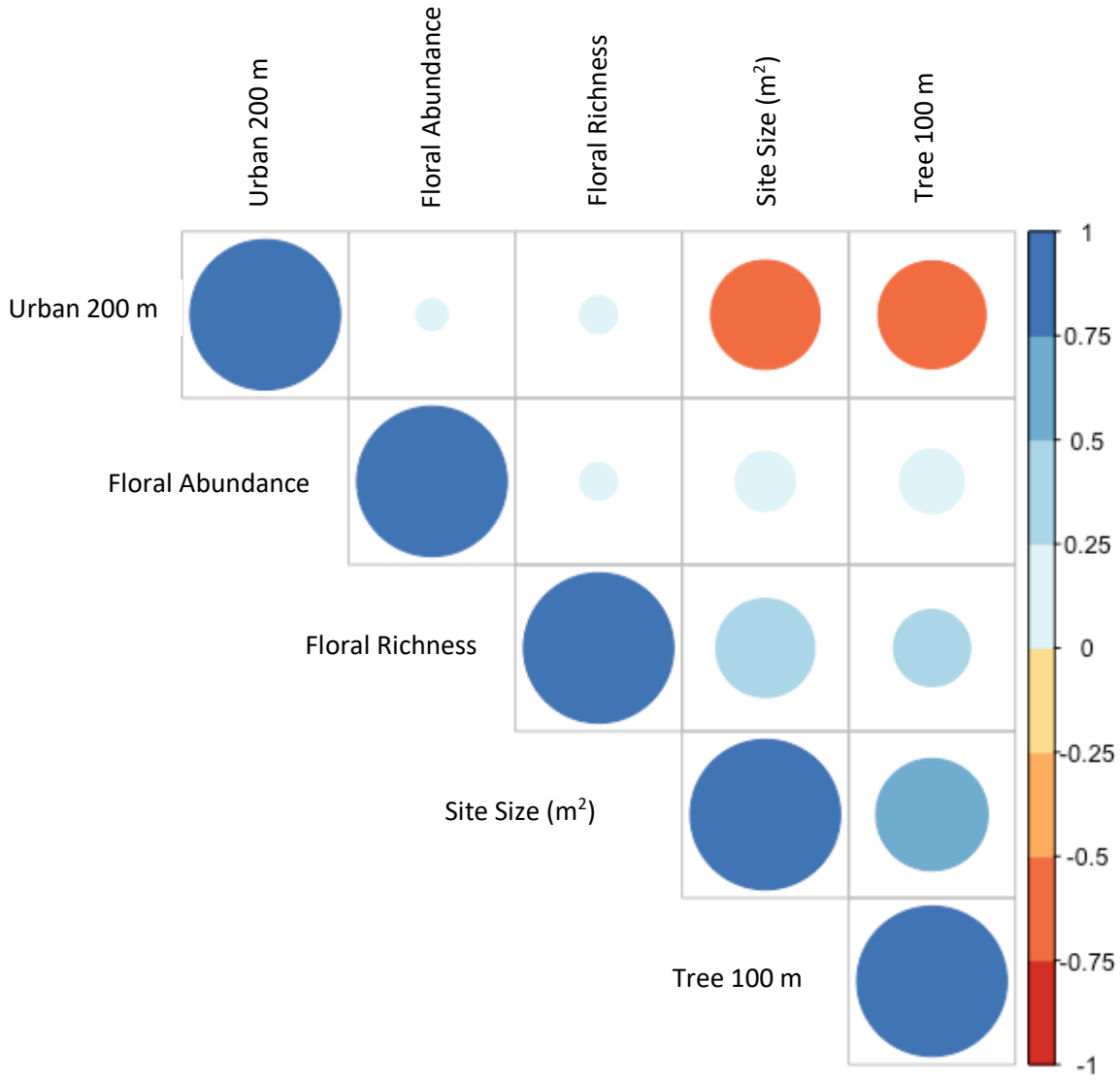

Figure 2.4. Correlation matrix for predictor variables used in bee diversity models

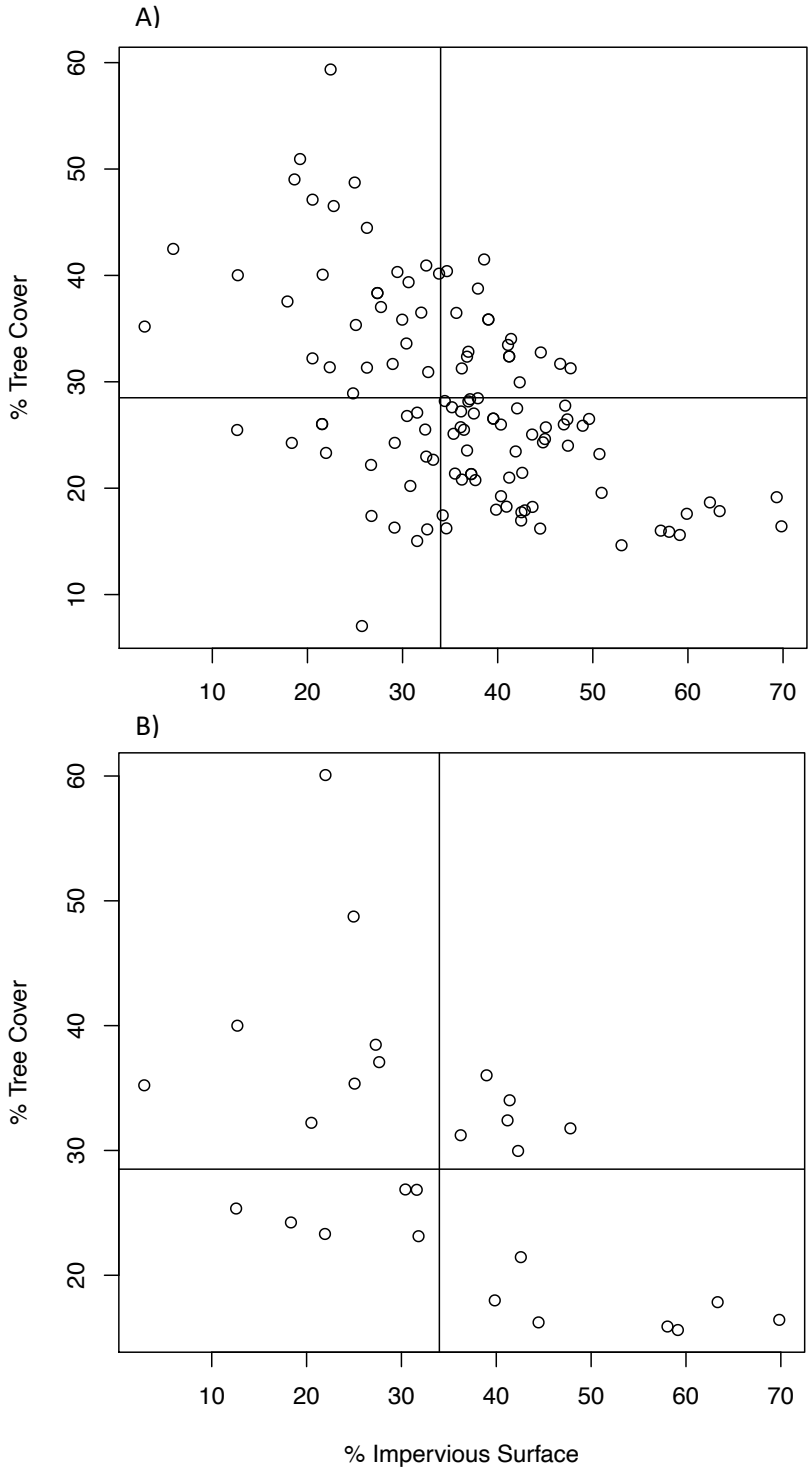

Figure 2.5. Illustration of the steps used in the site selection process. A) Volunteer sites that met the criteria were evaluated on their standing within the two gradients. B) Selected sites and their standing within the two gradients.

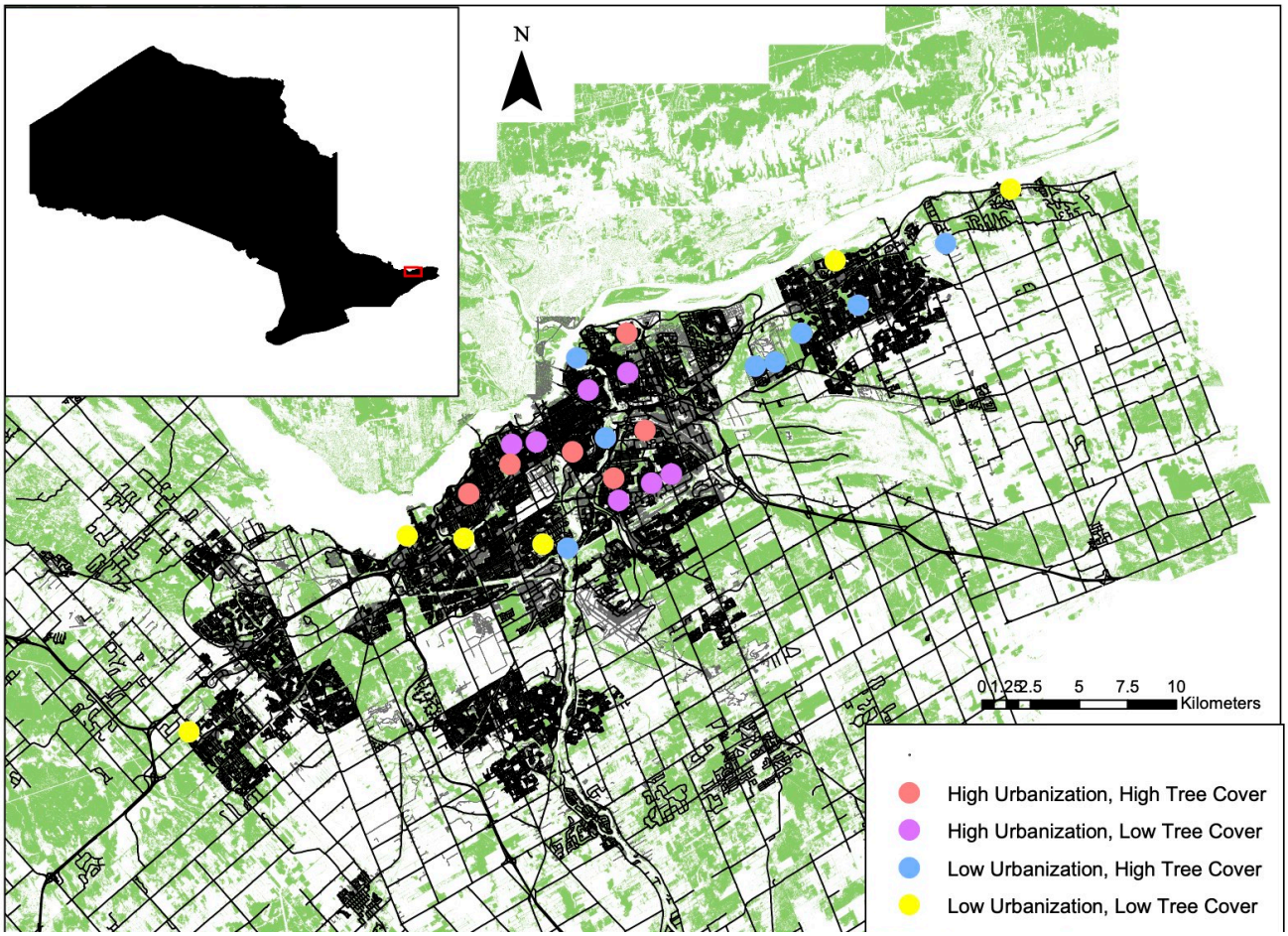

Figure 2.6. Map of the study region in Ottawa, Ontario, Canada. Sites shown are placed into four categories based on the surrounding percent urbanization and tree cover at 500m. Green colouring represents tree cover, while black colouring represents impervious surface area.

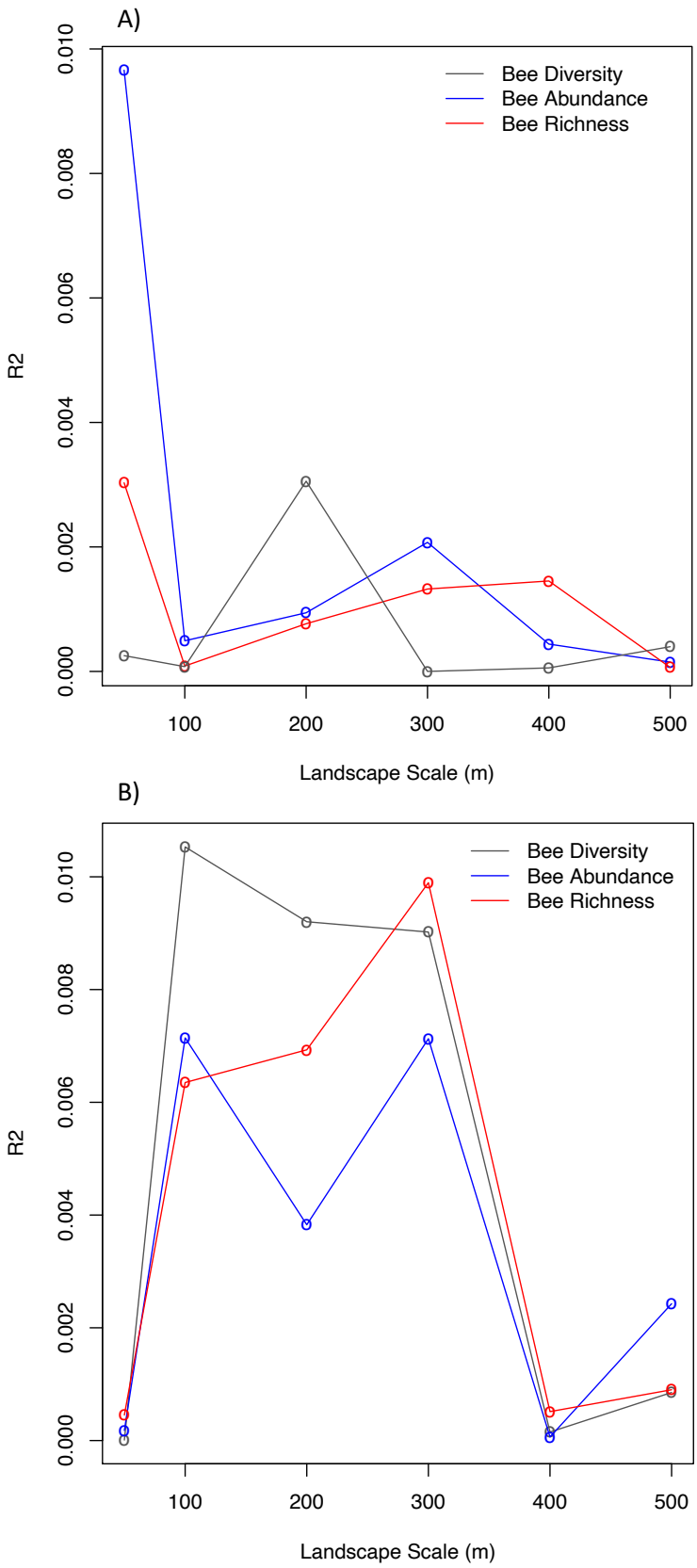

Figure 2.7. R^2 values plotted against landscape scale for bee abundance, richness, and diversity for the response gradients of A) impervious surface and B) tree cover

Appendix A: Site Volunteer Survey

Figure A1. An example of the survey study volunteers were required to fill out

Contact Info:

Name: _____

Address: _____

Area of Ottawa (ex. Kanata, Vanier): _____

Phone #:_____ Email:_____

How would you like to be contacted? TEXT, CALL, EMAIL

Yard Info:

You would like us to survey your (choose one): FRONT YARD or BACKYARD

What is the approximate size of your yard? SMALL, MEDIUM, LARGE

How many trees are present in your yard? _____

Does your yard have approximately 50% green space (grass or garden): YES or NO

Do you use pesticides on weeds? YES or NO

Do you have a family dog? YES or NO

What level of weeding does your yard receive? INFREQUENT (Less than once a month)

INTERMEDIATE (at least once a month), FREQUENT (at least twice a month)

What level of gardening experience is reflected in your yard? LOW, MEDIUM, HIGH

Additional Comments or Questions: _____

_____

_____

Appendix B: Site Percentages of Landscape Gradients

Table B1. Percent area of impervious surface at each spatial scale for the selected study sites.

Site Number	Spatial Scale					
	50m (%)	100m (%)	200m (%)	300m (%)	400m (%)	500m (%)
S1	25.6	31.6	22.9	21.4	20.7	18.4
S2	49.9	58.8	61.9	58.1	57.9	58.0
S3	22.3	27.4	25.0	26.3	29.0	31.8
S4	44.3	44.2	35.0	30.4	26.6	25.1
S5	45.1	46.2	44.0	39.0	40.8	42.6
S6	45.3	41.9	43.5	42.5	39.8	39.8
S7	44.0	38.5	39.2	36.9	35.6	36.2
S8	43.1	41.6	39.7	39.8	39.8	41.2
S9	52.8	60.3	59.9	64.0	64.4	63.3
S10	65.7	67.8	66.8	68.8	69.9	69.8
S11	36.4	38.3	37.3	38.0	40.7	39.0
S12	58.4	59.6	57.9	52.2	58.2	59.1
S13	48.2	44.0	49.3	48.2	49.0	47.8
S14	41.4	29.7	22.2	23.4	25.7	27.3
S15	29.0	41.1	40.0	32.5	29.0	27.7
S16	1.60	1.30	2.20	2.80	3.50	2.90
S17	36.8	33.4	37.4	44.6	43.7	44.5
S18	33.6	38.4	44.0	39.2	35.0	31.6
S19	43.2	44.5	45.5	44.0	42.6	42.3
S20	21.5	16.5	17.9	15.0	14.0	12.6
S21	45.2	41.9	31.7	25.8	23.9	22.0
S22	18.9	27.1	21.2	15.8	12.0	12.7
S23	56.2	58.0	39.4	29.1	26.4	20.5
S24	34.4	34.2	29.3	25.9	22.4	21.9
S25	43.2	43.7	38.5	30.9	27.5	25.0
S26	37.3	36.9	40.0	37.8	34.3	30.4
S27	44.0	46.7	45.4	45.2	43.9	41.4

Table B2. Percent area of tree cover at each spatial scale for the selected study sites

Site Number	Spatial Scale					
	50m (%)	**100m (%)**	**200m (%)**	**300m (%)**	**400m (%)**	**500m (%)**
S1	3.30	5.40	14.4	18.3	21.6	24.2
S2	25.6	21.1	15.7	16.4	16.1	15.9
S3	21.2	21.9	25.4	25.3	23.3	23.1
S4	30.6	30.9	33.4	28.7	35.3	35.4
S5	34.8	26.9	26.3	23.6	21.7	21.5
S6	29.1	28.4	21.9	19.2	18.7	18.0
S7	33.7	36.8	32.5	31.0	31.3	31.2
S8	26.1	31.3	36.6	34.1	32.1	32.4
S9	17.6	19.8	19.7	18.7	19.0	17.8
S10	21.3	17.7	19.5	18.4	16.3	16.4
S11	56.1	52.6	46.3	42.9	38.3	36.0
S12	16.5	17.9	18.5	17.0	15.6	15.6
S13	23.9	28.6	27.5	30.4	31.2	31.8
S14	20.8	38.0	48.1	45.6	41.2	38.5
S15	34.4	24.3	31.0	40.7	41.2	37.1
S16	38.6	78.9	72.2	51.5	41.5	35.2
S17	30.3	28.7	28.0	21.4	17.4	16.2
S18	34.4	28.2	23.2	26.0	25.9	26.8
S19	39.6	38.7	32.5	32.6	31.4	30.0
S20	30.4	22.6	19.4	24.2	23.5	25.3
S21	27.9	31.0	44.8	54.9	59.0	60.1
S22	39.9	43.5	42.9	46	40.5	40.0
S23	28.8	28.1	32.1	30.9	32.9	32.2
S24	37.6	36	41.4	29.8	27.0	23.3
S25	24.8	25.1	27.7	36.1	42.0	48.7
S26	36.7	26.0	23.9	22.3	22.4	26.9
S27	36.3	34.3	33.9	36.0	34.1	34.0

Appendix C: Floral Resources

Floral resources surveyed in the gardens were classified according to 1) pollination strategy (pollinated by bees or by other insects/animals); and 2) their attractiveness level to a bee (high, medium or low) (Table C1). Attractiveness level was determined from the primary literature. A plant was ascribed high attractiveness if it met at least one of the following criteria: 1) it commonly appears in extension catalogues for pollinator gardens (City of Guelph, 2020; CVC, 2017; NRCS, 2011; PPC, 2017); 2) empirical studies show high visitation rates; 3) they are known to be pollinated by at least five or more different bee genera. Medium attractiveness was characterized by 1) plants that are considered to be attractive to bees but are not commonly advised in extension literature; 2) studies that show medium levels of visitation rates; and 3) are pollinated by 3-4 bee genera. A plant that had low attractiveness was characterized by 1) plants are only briefly mentioned that they are visited by bees; 2) studies show low visitation rates in the field; 3) plants are only visited by 1-2 bee genera. Attractiveness level was only collected for flowers that were identified to the species level (see Table C2 for the breakdown of references by plant family).

Table C1. Flower species/morphospecies found during residential yard

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Actinidia kolomikta	Actinidiaceae	B	L
Sambucus racemosa	Adoxaceae	B	L
Viburnum opulus	Adoxaceae	B	H
Viburnum species1	Adoxaceae	B	-
Celosia argentea	Amaranthaceae	B	L
Allium canadense	Amaryllidaceae	B	H
Allium giganteum	Amaryllidaceae	B	H
Allium schoenoprasum	Amaryllidaceae	B	H
Allium species1	Amaryllidaceae	B	-
Narcissus species1	Amaryllidaceae	B	-
Cotinus coggygria	Anacardiaceae	B	L
Aegopodium podagraria	Apiaceae	B	L
Anethum graveolens	Apiaceae	B	H
Astrantia major	Apiaceae	B	L
Coriandrum sativum	Apiaceae	B	M
Daucus carota	Apiaceae	B	M
Foeniculum vulgare	Apiaceae	B	H
Amsonia species1	Apocynaceae	B	-
Asclepias incarnata	Apocynaceae	B	H
Asclepias syriaca	Apocynaceae	B	H
Asclepias tuberosa	Apocynaceae	B	H
Catharanthus roseus	Apocynaceae	B	L
Mandevilla sanderi	Apocynaceae	B	L
Vinca minor	Apocynaceae	B	M
Ilex aquifolium	Aquifoliaceae	B	M
Convallaria majalis	Asparagaceae	B	L
Hosta species1	Asparagaceae	B	-
Hyacinthus orientalis	Asparagaceae	B	L
Muscari botryoides	Asparagaceae	B	M
Ornithogalum species1	Asparagaceae	B	-
Polygonatum species1	Asparagaceae	B	-
Polygonum convolvulus	Asparagaceae	B	M
Polygonum species1	Asparagaceae	B	-
Scilla species1	Asparagaceae	B	-
Hemerocallis species1	Asphodelaceae	B	-
Achillea species1	Asteraceae	B	-
Achillea species2	Asteraceae	B	-
Achillea species3	Asteraceae	B	-

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Aconitum species1	Asteraceae	B	-
Ageratina altissima	Asteraceae	B	L
Anthemis arvensis	Asteraceae	B	H
Arctium minus	Asteraceae	B	L
Arctium species1	Asteraceae	B	-
Arctium species2	Asteraceae	B	-
Argyranthemum species1	Asteraceae	B	-
Aster species1	Asteraceae	B	-
Aster species2	Asteraceae	B	-
Bellis perennis	Asteraceae	B	L
Bidens species1	Asteraceae	B	-
Calendula officinalis	Asteraceae	B	H
Catananche caerulea	Asteraceae	B	L
Centaurea species1	Asteraceae	B	-
Centaurea species2	Asteraceae	B	-
Centaurea species3	Asteraceae	B	-
Chrysogonum virginianum	Asteraceae	B	M
Cirsium species1	Asteraceae	B	-
Cirsium species2	Asteraceae	B	-
Cirsium species3	Asteraceae	B	-
Coreopsis grandiflora	Asteraceae	B	H
Coreopsis lanceolata	Asteraceae	B	H
Coreopsis species1	Asteraceae	B	-
Coreopsis verticillata	Asteraceae	B	H
Cosmos bipinnatus	Asteraceae	B	H
Dahlia pinnata	Asteraceae	B	H
Dahlia species1	Asteraceae	B	-
Echinacea purpurea	Asteraceae	B	H
Echinops species1	Asteraceae	B	-
Erigeron annuus	Asteraceae	B	H
Erigeron canadensis	Asteraceae	B	H
Erigeron philadelphicus	Asteraceae	B	H
Eutrochium species1	Asteraceae	B	-
Gaillardia aristata	Asteraceae	B	H
Gerbera species1	Asteraceae	B	-
Helianthus species1	Asteraceae	B	-
Helianthus species2	Asteraceae	B	-
Helianthus strumosus	Asteraceae	B	H

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Heliopsis helianthoides	Asteraceae	B	H
Lactuca species1	Asteraceae	B	-
Lactuca species2	Asteraceae	B	-
Layia species1	Asteraceae	B	-
Leucanthemum × superbum	Asteraceae	B	H
Leucanthemum vulgare	Asteraceae	B	M
Liatris species1	Asteraceae	B	-
Liatris spicata	Asteraceae	B	H
Ligularia species1	Asteraceae	B	-
Matricaria recutita	Asteraceae	B	M
Pilosella caespitosa	Asteraceae	B	L
Rudbeckia fulgida	Asteraceae	B	H
Rudbeckia hirta	Asteraceae	B	H
Rudbeckia laciniata	Asteraceae	B	H
Rudbeckia species1	Asteraceae	B	-
Rudbeckia triloba	Asteraceae	B	H
Solidago species1	Asteraceae	B	-
Solidago species2	Asteraceae	B	-
Sonchus species1	Asteraceae	B	-
Sonchus species2	Asteraceae	B	-
Symphyotrichum novi-belgii	Asteraceae	B	H
Tagetes lucida	Asteraceae	B	L
Tagetes species1	Asteraceae	B	-
Tanacetum parthenium	Asteraceae	B	L
Tanacetum vulgare	Asteraceae	B	M
Taraxacum species1	Asteraceae	B	-
Zinna species1	Asteraceae	B	-
Zinnia elegans	Asteraceae	B	M
Impatiens capensis	Balsaminaceae	B	L
Impatiens species1	Balsaminaceae	B	-
Begonia semperflorens	Begoniaceae	B	L
Begonia species1	Begoniaceae	B	-
Begonia x intermedia	Begoniaceae	B	L
Epimedium x rubrum	Berberidaceae	B	U
Epimedium x versicolour	Berberidaceae	B	U
Borago officinalis	Boraginaceae	B	H
Brunnera macrophylla	Boraginaceae	B	U
Cynoglossum amabile	Boraginaceae	B	U

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Hydrophyllum virginianum	Boraginaceae	B	H
Mertensia virginica	Boraginaceae	B	H
Myosotis species1	Boraginaceae	B	-
Nemophila maculata	Boraginaceae	B	M
Phacelia tanacetifolia	Boraginaceae	B	H
Pulmonaria species1	Boraginaceae	B	-
Symphytum officinale	Boraginaceae	B	L
Alliaria petiolata	Brassicaceae	B	L
Arabis species1	Brassicaceae	B	-
Arabis x arendsii	Brassicaceae	B	U
Aubrieta species1	Brassicaceae	B	-
Barbarea species1	Brassicaceae	B	-
Barbarea species2	Brassicaceae	B	-
Berteroa incana	Brassicaceae	B	L
Brassica oleracea	Brassicaceae	B	L
Brassica rapa	Brassicaceae	B	M
Brassica species1	Brassicaceae	B	-
Brassica species2	Brassicaceae	B	-
Capsella bursa-pastoris	Brassicaceae	B	L
Diplotaxis species1	Brassicaceae	B	-
Erysimum species1	Brassicaceae	B	-
Hesperis matronalis	Brassicaceae	B	L
Raphanus raphanistrum	Brassicaceae	B	M
Thlaspi arvense	Brassicaceae	B	H
Pachysandra terminalis	Buxaceae	B	L
Campanula species1	Campanulaceae	B	-
Campanula species2	Campanulaceae	B	-
Campanula species3	Campanulaceae	B	-
Campanula species4	Campanulaceae	B	-
Campanula species5	Campanulaceae	B	-
Lobelia cardinalis	Campanulaceae	B	H
Lobelia erinus	Campanulaceae	B	H
Lobelia inflata	Campanulaceae	B	H
Lobelia species1	Campanulaceae	B	-
Lobularia maritima	Campanulaceae	B	H
Platycodon grandiflorus	Campanulaceae	B	L
Linnaea amabilis	Caprifoliaceae	B	M
Lonicera japonica	Caprifoliaceae	B	H

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Lonicera species1	Caprifoliaceae	B	-
Lonicera tatarica	Caprifoliaceae	B	L
Lonicera x bella	Caprifoliaceae	B	L
Symphoricarpos albus	Caprifoliaceae	B	H
Valeriana officinalis	Caprifoliaceae	B	H
Weigela species1	Caprifoliaceae	B	-
Arenaria species1	Caryophyllaceae	B	-
Armeria maritima	Caryophyllaceae	B	H
Armeria species1	Caryophyllaceae	B	-
Cerastium fontanum	Caryophyllaceae	B	L
Cerastium species1	Caryophyllaceae	B	-
Cerastium tomentosum	Caryophyllaceae	B	U
Lychnis chalcedonica	Caryophyllaceae	B	H
Sagina species1	Caryophyllaceae	B	-
Silene coronaria	Caryophyllaceae	B	U
Silene vulgaris	Caryophyllaceae	B	U
Spergularia spp	Caryophyllaceae	B	-
Stellaria species1	Caryophyllaceae	B	-
Stellaria species2	Caryophyllaceae	B	-
Delosperma species1	Caryophyllales	B	-
Dianthus armeria	Caryophyllales	B	L
Dianthus deltoides	Caryophyllales	B	L
Dianthus species1	Caryophyllales	B	-
Euonymus alatus	Celastraceae	B	M
Cleome hassleriana	Cleomaceae	B	L
Tradescantia species1	Commelinaceae	B	-
Tradescantia species2	Commelinaceae	B	-
Calystegia sepium	Convolvulaceae	B	L
Ipomoea purpurea	Convolvulaceae	B	L
Cornus canadensis	Cornaceae	B	H
Cornus species1	Cornaceae	B	-
Cornus species2	Cornaceae	B	-
Hylotelephium species1	Crassulaceae	B	-
Sedum acre	Crassulaceae	B	H
Sedum cauticola	Crassulaceae	B	H
Sedum kamschaticum	Crassulaceae	B	H
Sedum kamtschaticum	Crassulaceae	B	H
Sedum reflexum	Crassulaceae	B	H

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Sedum species1	Crassulaceae	B	-
Sedum species2	Crassulaceae	B	-
Sedum telephium	Crassulaceae	B	H
Cucumis sativus	Cucurbitaceae	B	L
Cucurbita pepo	Cucurbitaceae	B	M
Cucurbita pepo var. cylindrica	Cucurbitaceae	B	M
Cucurbita pepo var. pepo	Cucurbitaceae	B	M
Rhododendron species1	Ericaceae	B	-
Euphorbia epithymoides	Euphorbiaceae	B	U
Euphorbia oblongata	Euphorbiaceae	B	U
Euphorbia species1	Euphorbiaceae	B	-
Baptisia australis	Fabaceae	B	H
Caragana arborescens	Fabaceae	B	L
Desmodium species1	Fabaceae	B	H
Lathyrus latifolius	Fabaceae	B	L
Lathyrus species1	Fabaceae	B	-
Lathyrus species2	Fabaceae	B	-
Medicago lupulina	Fabaceae	B	L
Melilotus alba	Fabaceae	B	M
Phaseolus vulgaris	Fabaceae	B	L
Pisum sativum	Fabaceae	B	L
Trifolium pratense	Fabaceae	B	M
Trifolium repens	Fabaceae	B	M
Vicia species1	Fabaceae	B	-
Quercus species1	Fagaceae	B	-
Geranium macrorrhizum	Geraniaceae	B	M
Geranium maculatum	Geraniaceae	B	H
Geranium pratense	Geraniaceae	B	H
Geranium sanguineum	Geraniaceae	B	M
Geranium x cantabrigiense	Geraniaceae	B	M
Pelargonium interspecific	Geraniaceae	B	L
Pelargonium species1	Geraniaceae	B	-
Ribes americanum	Grossulariaceae	B	M
Ribes aureum	Grossulariaceae	B	H
Ribes species1	Grossulariaceae	B	-
Hydrangea quercifolia	Hydrangeaceae	B	L
Hydrangea species1	Hydrangeaceae	B	-
Philadelphus x virginalis	Hydrangeaceae	B	M

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Hypericum perforatum	Hypericaceae	B	H
Crocosmia species1	Iridaceae	B	-
Gladiolus species1	Iridaceae	B	-
Iris species1	Iridaceae	B	-
Sisyrinchium species1	Iridaceae	B	-
Ajuga reptans	Lamiaceae	B	L
Glechoma hederacea	Lamiaceae	B	M
Hyssopus officinalis	Lamiaceae	B	H
Lamium galeobdolon	Lamiaceae	B	L
Lamium maculatum	Lamiaceae	B	L
Lamium purpureum	Lamiaceae	B	L
Lavandula species1	Lamiaceae	B	-
Lavandula stoechas	Lamiaceae	B	H
Leonurus cardiaca	Lamiaceae	B	L
Mentha species1	Lamiaceae	B	-
Mentha species2	Lamiaceae	B	-
Mentha species3	Lamiaceae	B	-
Monarda didyma	Lamiaceae	B	H
Monarda media	Lamiaceae	B	H
Monarda species1	Lamiaceae	B	-
Monarda species2	Lamiaceae	B	-
Nepeta species1	Lamiaceae	B	-
Ocimum basilicum	Lamiaceae	B	H
Origanum vulgare	Lamiaceae	B	H
Perovskia species1	Lamiaceae	B	-
Plectranthus species1	Lamiaceae	B	-
Prunella vulgaris	Lamiaceae	B	L
Salvia species1	Lamiaceae	B	-
Salvia x sylvestris	Lamiaceae	B	H
Stachys byzantina	Lamiaceae	B	H
Stachys monieri	Lamiaceae	B	H
Thymus serpyllum	Lamiaceae	B	U
Thymus species1	Lamiaceae	B	-
Lilium species1	Liliaceae	B	-
Tulipa × gesneriana	Liliaceae	B	M
Tulipa sylvestris	Liliaceae	B	M
Linum usitatissimum	Linaceae	B	L
Lythrum salicaria	Lythraceae	B	L

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Magnolia species1	Magnoliaceae	B	-
Alcea rosea	Malvaceae	B	L
Althaea officinalis	Malvaceae	B	L
Hibiscus moscheutos	Malvaceae	B	L
Hibiscus species1	Malvaceae	B	-
Lavatera species1	Malvaceae	B	-
Malva alcea	Malvaceae	B	H
Malva neglecta	Malvaceae	B	L
Tilia species1	Malvaceae	B	-
Trillium erectum	Melanthiaceae	B	H
Trillium grandiflorum	Melanthiaceae	B	H
Trillium species1	Melanthiaceae	B	-
Lotus species1	Nelumbonaceae	B	-
Forsythia species1	Oleaceae	B	-
Jasminum species1	Oleaceae	B	-
Syringa meyeri	Oleaceae	B	M
Syringa reticulata	Oleaceae	B	L
Syringa vulgaris	Oleaceae	B	H
Circaea canadensis	Onagraceae	B	U
Epilobium species1	Onagraceae	B	-
Oenothera species1	Onagraceae	B	-
Oxalis species1	Oxalidaceae	B	-
Oxalis species2	Oxalidaceae	B	-
Paeonia species1	Paeoniaceae	B	-
Chelidonium majus	Papaveraceae	B	M
Eschscholzia californica	Papaveraceae	B	M
Lamprocapnos spectabilis	Papaveraceae	B	L
Papaver orientale	Papaveraceae	B	L
Papaver rupifragum	Papaveraceae	B	U
Papaver species1	Papaveraceae	B	-
Antirrhinum species1	Plantaginaceae	B	-
Digitalis lutea	Plantaginaceae	B	U
Digitalis purpurea	Plantaginaceae	B	L
Penstemon digitalis	Plantaginaceae	B	H
Penstemon species1	Plantaginaceae	B	-
Penstemon species2	Plantaginaceae	B	-
Veronica arvensis	Plantaginaceae	B	L
Veronica chamaedrys	Plantaginaceae	B	L

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Veronica longifolia	Plantaginaceae	B	L
Veronica peregrina	Plantaginaceae	B	L
Veronica persica	Plantaginaceae	B	L
Veronica serpyllifolia	Plantaginaceae	B	L
Veronica spicata	Plantaginaceae	B	H
Veronicastrum virginicum	Plantaginaceae	B	H
Limonium species1	Plumbaginaceae	B	-
Phlox pilosa	Polemoniaceae	B	H
Phlox species1	Polemoniaceae	B	-
Phlox species2	Polemoniaceae	B	-
Phlox subulata	Polemoniaceae	B	H
Polemonium caeruleum	Polemoniaceae	B	H
Fallopia species1	Polygonaceae	B	-
Persicaria species1	Polygonaceae	B	-
Portulaca grandiflora	Portulacaceae	B	M
Lysimachia ciliata	Primulaceae	B	L
Lysimachia clethroides	Primulaceae	B	L
Lysimachia nummularia	Primulaceae	B	L
Lysimachia punctata	Primulaceae	B	L
Lysimachia species1	Primulaceae	B	-
Primula vulgaris	Primulaceae	B	L
Anemone blanda	Ranunculaceae	B	H
Anemone canadensis	Ranunculaceae	B	H
Anemone coronaria	Ranunculaceae	B	H
Anemone hupehensis	Ranunculaceae	B	L
Aquilegia canadensis	Ranunculaceae	B	H
Aquilegia species1	Ranunculaceae	B	-
Clematis species1	Ranunculaceae	B	-
Clematis species2	Ranunculaceae	B	-
Delphinium species2	Ranunculaceae	B	-
Helleborus orientalis	Ranunculaceae	B	U
Pulsatilla species1	Ranunculaceae	B	-
Thalictrum species1	Ranunculaceae	B	-
Trollius europaeus	Ranunculaceae	O	N
Rhamnus cathartica	Rhamnaceae	B	M
Alchemilla species1	Rosaceae	B	-
Amelanchier species1	Rosaceae	B	-
Aruncus dioicus	Rosaceae	B	M

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Crataegus species1	Rosaceae	B	-
Dryas species1	Rosaceae	B	-
Filipendula rubra	Rosaceae	B	M
Fragaria × ananassa	Rosaceae	B	M
Fragaria vesca	Rosaceae	B	H
Geum species1	Rosaceae	B	-
Geum species2	Rosaceae	B	-
Kerria japonica	Rosaceae	B	L
Malus domestica	Rosaceae	B	L
Malus species1	Rosaceae	B	-
Physocarpus opulifolius	Rosaceae	B	H
Potentilla species1	Rosaceae	B	-
Potentilla species2	Rosaceae	B	-
Prunus × cistena	Rosaceae	B	H
Prunus cerasus	Rosaceae	B	H
Prunus species1	Rosaceae	B	-
Prunus virginiana	Rosaceae	B	H
Pyrus species1	Rosaceae	B	-
Rosa rubiginosa	Rosaceae	B	H
Rosa rugosa	Rosaceae	B	H
Rosa species1	Rosaceae	B	-
Rubus idaeus	Rosaceae	B	H
Rubus odoratus	Rosaceae	B	H
Sorbaria sorbifolia	Rosaceae	B	M
Spiraea betulifolia	Rosaceae	B	H
Spiraea cantoniensis	Rosaceae	B	H
Spiraea japonica	Rosaceae	B	H
Spiraea species1	Rosaceae	B	-
Galium odoratum	Rubiaceae	B	U
Galium palustre	Rubiaceae	B	L
Galium species1	Rubiaceae	B	-
Houstonia caerulea	Rubiaceae	B	M
Calamansi species1	Rutaceae	B	-
Acer negundo	Sapindaceae	B	L
Acer rubrum	Sapindaceae	B	L
Acer species1	Sapindaceae	B	-
Aesculus x carnea	Sapindaceae	B	H
Houttuynia cordata	Saururaceae	B	L

Species	Family	Pollination Strategy[1]	Attractiveness Level[2]
Astilbe species1	Saxifragaceae	B	-
Bergenia crassifolia	Saxifragaceae	B	L
Heuchera sanguinea	Saxifragaceae	B	H
Heuchera species1	Saxifragaceae	B	-
Heucherella species1	Saxifragaceae	B	-
Rodgersia pinnata	Saxifragaceae	B	L
Rodgersia species1	Saxifragaceae	B	-
Tiarella species1	Saxifragaceae	B	-
Sutera cordata	Scrophulariaceae	B	L
Verbascum phoeniceum	Scrophulariaceae	B	L
Capsicum annuum	Solanaceae	B	L
Capsicum species1	Solanaceae	B	-
Capsicum species2	Solanaceae	B	-
Capsicum species3	Solanaceae	B	-
Datura inoxia	Solanaceae	B	U
Petunia species1	Solanaceae	B	-
Physalis alkekengi	Solanaceae	B	L
Solanum lycopersicum	Solanaceae	B	L
Solanum species1	Solanaceae	B	-
Solanum species2	Solanaceae	B	-
Tropaeolum species1	Tropaeolaceae	B	-
Verbena hastata	Verbenaceae	B	H
Verbena hybrida	Verbenaceae	B	H
Verbena species1	Verbenaceae	B	-
Viola blanda	Violaceae	B	H
Viola species1	Violaceae	B	-
Viola tricolor	Violaceae	B	L

[1] Pollination strategy: each species is classified as bee pollinated (B) or other pollinated (O)

[2] Attractiveness level: flowers were categorized by bee attractiveness level high (H), medium (M), low (L), or unknown (U)

- hyphen indicates a plant identified to morphospecies where attractiveness level could not be determined

Table C2. References used in Table C1, broken down by plant family. A hyphen indicates recorded plant was not ascribed an attractiveness level

Flower Family	**References Used**
Actinidiaceae	(Ferguson, 2016)
Adoxaceae	(Bobiwash et al., 2018)
Amaranthaceae	(Aluri & Chappidi, 2018)
Amaryllidaceae	Iowa
Anacardiaceae	(Mészáros & Józan, 2018)
Apiaceae	(Ahrné et al., 2009; City of Guelph, 2020; Lysenkov & Galinskaya, 2017; Ranjitha et al., 2019; Singh et al., 2017)
Apocynaceae	(Hassan, 2018; Willmer & Finlayson, 2014)
Aquifoliaceae	(Ollerton et al., 2016)
Asparagaceae	(Bożek, 2019; Odintsova & Fishchuk, 2017; Rollings & Goulson, 2019; Stubbs, 1992)
Asteraceae	(Baker & Baker, 1982; Beattie, 1972; Bobadoye et al., 2016; City of Guelph, 2020; Dow, 2019; Harrington, 2019; Jiju et al., 2017; Martins et al., 2017; Nicholls & Altieri, 2013; NRCS, 2011; Özbek & Terzo, 2016; PPC, 2017; Rollings & Goulson, 2019; Wróblewska et al., 2016; Xerces Society, 2017)
Balsaminaceae	(McDonald & Caruso, 2019)
Begoniaceae	(Kraai, 1962)
Berberidaceae	-
Boraginaceae	(Majetic et al., 2009)
Brassicaceae	(Ara et al., 2019; City of Guelph, 2020; Conner & Rush, 1996; Majetic et al., 2009; Mészáros & Jozan, 2018; Stanley et al., 2017)
Buxaceae	(Nagamitsu et al., 2010)
Campanulaceae	(City of Guelph, 2020; Nikkeshi et al., 2015; NRCS, 2011)
Caprifoliaceae	(Bergström et al., 2018; City of Guelph, 2020; Rollings & Goulson, 2019)
Caryophyllaceae	(City of Guelph, 2020; Harrington, 2019; Rollings & Goulson, 2019; Wyatt, 1986)
Caryophyllales	(City of Guelph, 2020; Rollings & Goulson, 2019)
Celastraceae	(Thompson & Knight, 2018)
Cleomaceae	(City of Guelph, 2020; Rollings & Goulson, 2019)
Commelinaceae	(NRCS, 2011)
Convolvulaceae	(Harrington, 2019; Paine et al., 2019)
Cornaceae	(NRCS, 2011)
Crassulaceae	(City of Guelph, 2020)
Cucurbitaceae	(Nikolova et al., 2019; Santos et al., 2008)
Ericaceae	-

Flower Family	References Used
Euphorbiaceae	-
Fabaceae	(Bosch et al., 1997; Colla & Dumesh, 2010; Free, 1966; Palmer-Jones et al., 1966; Polowick et al., 2002; Richards, 2003; Rodet & Grossa, 1998; Rollings & Goulson, 2019; Tsevegmid et al., 2018; Turkington & Cavers, 1979)
Geraniaceae	(CVC, 2017; Masierowska et al., 2018; Rollings & Goulson, 2019; Struck & Van der Walt, 1996; Willmer & Finlayson, 2014)
Grossulariaceae	(City of Guelph, 2020; Stubbs et al., 1992)
Hydrangeaceae	(Knudsen et al., 1993; Mach & Potter, 2017)
Hypericaceae	(City of Guelph, 2020)
Iridaceae	(PPC, 2017)
Lamiaceae	(Affek, 2018; City of Guelph, 2020; Cussans et al., 2010; Kuriya et al., 2015; Mačukanović-Jocić et al., 2004; Ouvrard et al., 2018; PPC, 2017; Rollings & Goulson, 2019; Shekari et al., 2018; Xerces Society, 2017)
Liliaceae	(Stubbs et al., 1992)
Linaceae	(Williams et al., 1991)
Lythraceae	(Grabas & Laverty, 1999)
Magnoliaceae	-
Malvaceae	(Ara et al., 2019; Davidson et al., 2020; Kudoh & Whigham, 1998; Paray et al., 2014; Rollings & Goulson, 2019)
Melanthiaceae	(PPC, 2017)
Nelumbonaceae	-
Oleaceae	(Denisow & Strzalkowska-Abramek 2013)
Onagraceae	-
Oxalidaceae	-
Paeoniaceae	-
Papaveraceae	(Anderson et al., 2019; Damerval & Becker, 2017; Humphreys & Gale, 1974; Rollings & Goulson, 2019; Tucker & Rehan, 2016)
Plantaginaceae	(Ara et al., 2018; Kampny, 1995; NRCS, 2011; Rollings & Goulson, 2019)
Plumbaginaceae	-
Polemoniaceae	(City of Guelph, 2020; NRCS, 2011; PPC, 2017)
Polygonaceae	-
Portulacaceae	(Keshtkar et al., 2015)
Primulaceae	(Boyd et al., 1990; Dötterl & Schäffler, 2007; Simpson & Neff, 1983)
Ranunculaceae	(City of Guelph, 2020; CVC, 2017; Kearns, C. A., & Inouye, 1994; Mach & Potter, 2019; NRCS, 2011; Rollings & Goulson, 2019)

Flower Family	References Used
Rhamnaceae	(Carreck et al., 1997)
Rosaceae	(Bennet et al., 2018; City of Guelph, 2020; Mach & Potter, 2019; MacInnis & Forrest, 2020; NRCS, 2011; PPC, 2017; Sikora et al., 2016; Stubbs et al., 1992; Ushimaru et al., 2008; Xerces Society, 2017)
Rubiaceae	(Somme et al., 2015; Robert Wyatt & Hellwig, 1979)
Rutaceae	-
Sapindaceae	(PPC, 2017; USDA, 2019)
Saururaceae	(Kato & Miura, 1996)
Saxifragaceae	(City of Guelph, 2020; Folk, 2015; Gilbert & Raven, 2014; NRCS, 2011; PPC, 2017)
Scrophulariaceae	(Rollings & Goulson, 2019)
Solanaceae	(Hogendoorn et al., 2010; Whitson & Manos, 2005)
Tropaeolaceae	(City of Guelph, 2020)
Verbenaceae	(NRCS, 2011)
Violaceae	(NRCS, 2011)

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Appendix D: Tree Species

Table D1. Tree species recorded at each site

Site	Common name	Species name	Abundance	Food Resource[1]
S1	Apple	*Malus domestica*	1	Y
	Cherry	*Prunus avium*	1	Y
	Oak	*Quercus spp*	1	Y
S2	Spruce	*Picea spp*	1	
	Royal Purple Smokebush	*Cotinus spp.*	1	Y
S3	Pine	*Pinus spp.*	1	
	Maple	*Acer spp.*	1	Y
S4	Silver Maple	*Acer saccharinum*	1	Y
	Chokecherry	*Prunus virginiana*	1	Y
	Serviceberry	*Amelanchier spp.*	1	Y
S5	Cedar	*Cedrus spp.*	1	
	Apple	*Malus domestica*	1	Y
	Elm	*Ulmus spp.*	1	Y
	Blue Spruce	*Picea pungen*	1	
	Pine	*Pinus spp.*	1	
S6	Maple	*Acer spp.*	1	Y
	Red Maple	*Acer rubrum*	1	Y
	Cedar	*Cedrus spp.*	1	
	Linden	*Tilia spp.*	1	Y
	Japanese Lilac	*Syringa reticulata*	1	Y
	Caragana	*Caragana spp.*	10	Y
S7	Crabapple	*Malus spp.*	1	Y
	Shagbark Hickory	*Carya ovata*	2	Y
S8	Ash	*Fraxinus spp.*	1	
S9	Japanese Lilac	*Syringa reticulata*	1	Y
	Elm	*Ulmus spp.*	1	Y
S10	Mountain Ash	*Sorbus spp.*	1	Y
	Maple	*Acer spp.*	1	Y
	Crabapple	*Malus spp.*	1	Y
S11	Maple	*Acer spp.*	2	Y
	Hemlock	*Tsuga spp.*	1	
S12	Pear	*Pyrus spp.*	2	Y
	Serviceberry	*Amelanchier spp.*	1	Y
	Nanaberry	*Rhus dentata*	2	Y
	Cherry	*Prunus cerasus*	1	Y

Site	Common name	Species name	Abundance	Food Resource[1]
S13	Red Chestnut	*Aesculus x carnea*	1	Y
	Cherry	*Prunus cerasus*	1	Y
	Hemlock	*Tsuga spp.*	1	
	Manitoba Maple	*Acer negundo*	1	Y
	Apple	*Malus domestica*	1	Y
S14	Maple	*Acer spp.*	3	Y
S14	Cherry	*Prunus cerasus*	1	Y
	Blue Spruce	*Picea pungens*	1	
	Honey Locust	*Gleditsia triacanthos*	1	Y
S15	Spruce	*Picea spp.*	1	
S16	Magnolia	*Magnolia spp.*	1	Y
	Crabapple	*Malus spp.*	1	Y
	Korean Lilac	*Syringa meyeri*	1	Y
	Blue Spruce	*Picea pungens*	2	
	Elm	*Ulmus spp.*	1	Y
	Hawthorne	*Crataegus spp.*	4	Y
	Chokecherry	*Prunus virginiana*	1	Y
	Forsythia	*Forsythia spp.*	1	Y
S17	Linden	*Tilia spp.*	1	Y
	Caragana	*Caragana spp.*	9	Y
S18	Japanese Lilac	*Syringa reticulata*	1	Y
	Apple	*Malus domestica*	1	Y
	Honey Locust	*Gleditsia triacanthos*	1	y
	Maple	*Acer spp.*	1	Y
	Chokecherry	*Prunus virginiana*	1	Y
	Hydrangea	*Hydrangea spp.*	1	Y
	Japanese Maple	*Acer palmatum*	1	Y
S19	Crabapple	*Malus spp.*	1	Y
	Hackberry	*Celtis occidentalis*	1	Y
	Chokecherry	*Prunus virginiana*	1	Y
S20	Red Maple	*Acer rubrum*	1	Y
	Crabapple	*Malus spp.*	1	Y
	Cedar	*Cedrus spp.*	1	
S21	Forsythia	*Forsythia spp.*	1	Y
	Maple	*Acer spp.*	1	Y
S22	Honey Locust	*Gleditsia triacanthos*	1	Y
	Pine	*Pinus spp.*	1	
	Cedar	*Cedrus spp.*	2	
	Oak	*Quercus spp.*	1	Y
	Spruce	*Picea spp.*	2	
	Silver Maple	*Acer saccharinum*	1	Y

Site	Common name	Species name	Abundance	Food Resource[1]
S22	Maple	*Acer spp.*	1	Y
	Korean Lilac	*Syringa meyeri*	1	Y
S23	Blue Spruce	*Picea pungens*	1	
	Birch	*Betula spp.*	1	Y
	Spruce	*Picea spp.*	1	
	Hemlock	*Tsuga spp.*	1	
	Serviceberry	*Amelanchier spp.*	1	Y
	Japanese Maple	*Acer palmatum*	1	Y
	Redbud	*Cercis canadensis*	1	Y
S24	Maple	*Acer spp.*	1	Y
	Manitoba Maple	*Acer negundo*	1	Y
	Chokecherry	*Prunus virginiana*	1	Y
S25	Hackberry	*Celtis spp.*	1	Y
	Korean Lilac	*Syringa meyeri*	1	Y
S26	Hickory	*Carya spp*	1	Y
	Maple	*Acer spp.*	2	Y
	Apple	*Malus domestica*	1	Y
	Manitoba Maple	*Acer negundo*	2	Y
	Weeping Mulberry	*Morus alba*	1	Y
S27	Maple	*Acer spp.*	6	Y

[1]Food Resource indicates if they provide food for wild bees, where (Y) represents yes, and a blank represents no

Appendix E: Bee Ecological Characteristics

Introduction

Unfortunately, for the majority of wild bee species, little or nothing is known about the factors that affect their presence in an area; although, some species have traits that appear to make them more adaptable to urban environments (Bartomeus et al., 2013; Williams et al., 2010). For example, the bee species most commonly found in city centres are polylectic, collecting pollen and nectar from a variety of plant species (Matteson et al., 2008). Contrarily, oligolectic bees, or bees that require pollen from specific plant species or families, tend to be scarcer than polylectic bees in disturbed environments due to their reliance on the presence of specific host plant(s) (Winfree et al., 2011). Studies have also shown that foraging range depends on bee body size, with larger bees (e.g., bumblebees) able to travel further to find floral resources than smaller bees (Gathmann & Tscharntke, 2002; Zurbuchen et al., 2010). Since the foraging range of most individual species is not currently known (Greenleaf et al., 2007), it is difficult to use this to predict how foraging range impacts bee species' ability to use particular environments. Nesting type is another ecological trait that can help to predict which bee species persist in urban areas. Studies have reported that cavity-nesting bees tend to increase as a total of all bee species along a gradient of impervious surfaces (Cane et al., 2006; Geslin et al., 2016). The infrastructure associated with urbanization can also create new environments for cavity-nesting to occur through added man-made structures (e.g., flower pots, cracks between bricks in homes and walls, fence posts) (Matteson et al., 2008; Neame et al., 2013). Despite improvements for cavity nesters, cities may lack patches of bare ground, which is essential for ground nesting bee establishment (Cane et al., 2006; Pardee & Philpott, 2014). Ground nesting

bees require well-drained, bare patches of loose soil in a sunny area (Everaars et al., 2018). Bare patches tend to be sparser in cities as opposed to agricultural and wild/semi-wild places (Cane et al., 2006; Threlfall et al., 2015). Ground nesting bees make up ~70% of native bee species in eastern Canada (Packer et al., 2007), so a lack of habitat could severely restrict the species diversity in urban areas. Urban ground-nesting bees also have to compete for nesting space with flower plantings and may be unable to establish in gardens or other potentially suitable places because of mulching and/or frequent irrigation of the soil (MacIvor, 2016). Urban areas can, therefore, act as a filter, eliminating species with specific traits (Grimm et al., 2008).

Materials & Methods

The key bee ecological characteristics were established following Matteson et al. (2008) (Table E1). Additional information was collected from the primary literature (Ascher et al., 2014; Birdshire, 2018; Blitzer et al., 2016; Gibbs, 2017; Giles & Ascher, 2006; Hobbs, 1956; Johnson, 1981; Lerman & Milam, 2016; Matteson et al., 2008; Michener, 2000; Müller, 2015; Normandin et al., 2017; Onuferko, 2018; Rozen & Favreau, 1968; Sheffield et al., 2008; Vickruck et al., 2011). Bees were classified by 1) origin (exotic or native); 2) floral specialization (oligolectic or polylectic), where oligolectic species were characterized as bees that had a pollen preference within a single plant family (Sedivy et al., 2008); 3) nesting guild (soil, cavity, soft wood, and pithy stems); 4) sociality (solitary, sub-social, and eusocial); 5) body size (small, medium, and large). Body sizes were obtained by measuring the intertegular (ITD) distance (distance between the wing bases) (Cane, 1987) of five female specimens using ImageJ software (Schneider et al., 2012). These were then averaged and placed into body size classes small (<1.8

mm), medium (1.9-3 mm) and large (>3.1 mm) (modified from Hinners et al., 2012). Intertegular distance is often used as a proxy for bee body size, which can affect foraging distances (Greenleaf et al., 2007). Only bumblebee workers were measured and grouped for body size classes. In cases where less than 5 females were caught, measured values were confirmed with body sizes shown in the literature (Lerman & Milam, 2016; Matteson et al., 2008). In species where only males were caught, ITD was not measured. To calculate the percentage of bees with specific life history traits, the total number of species with that trait were divided over the total number of species collected.

Results

The origins of the bees differed across sites. Out of the 98 species/morphospecies collected, only 9% are exotic and the rest are native to North America. Exotic bees included species like *Apis mellifera*, *Anthidium* spp. and *Chelostoma* spp. Similar results were found for pollen specificity, with 9% of species being oligolectic and the rest being polylectic (as defined by Sedivy et al., 2008). The majority of collected species (50%) nested in soil, however, this is below the expected number of ground nesting bee species (70%). The remaining species were cavity nesters (30%), pithy stems nesters (4%), wood nesters (1%), or were cleptoparasitic on other bee nests (15%). In terms of sociality, 64% of bees were solitary, 10% eusocial, and 4% sub-social, with the remaining bees having undetermined sociality due to a lack of species level identification. The body sizes of the collected bees were relatively evenly distributed between small (24%), medium (29%), and large (31%). The most commonly found species were from the family were Halictidae (34%), Apidae (22%), Megachilidae (20%), followed by Andrenidae (16%).

Colletidae only accounted for 7% of the family collections, probably because bees in this family are known to exhibit low attraction rates to pan traps (Giles & Ascher, 2006).

Discussion

Urban areas have previously been shown to support a wide diversity of bee species worldwide. In Montreal and Quebec City, QC (Normandin et al., 2017) 177 and 153 species were reported respectively (out of 365 known to the province, Ascher & Pickering, 2020); In Lyon, France (Fortel et al., 2014) 291 species were found (out of 912 in France, Leonhardt et al., 2013), while in New York, USA, 54 (out of 416 for NY State, Cornell University, 2020) (Matteson et al., 2008). I captured a total of 98 species (including morphospecies) in the City of Ottawa, Ontario. While estimates for how many species occur in Ottawa are not comprehensive, there are at least 350 bee species known to the area (City of Toronto, 2016). Overall, these studies demonstrate that anthropogenic habitats can support a diverse community of wild bees.

Certain ecological traits may allow some species to better persist in urban systems. For example, a generalist diet is thought to be advantageous for bee survival in urban areas (Winfree et al., 2011). Oligolectic bees rely on a specific plant species or group for pollen collection and may be food limited if the plant is difficult to find (Biesmeijer, 2006). This is consistent with my findings as the majority of bees sampled were polylectic; however, it is important to note that the majority of bees, in general, display no pollen preference (Ascher & Pickering, 2020). Bee body size is another trait that impacts a bee's success in an urban area. Bees with smaller body sizes may be more impacted by habitat loss than larger bodied bees, as they have a more limited ability to fly between food patches (Bommarco et al., 2010). Despite this, I found a relatively equal number of bees across small, medium and large body sizes.

A bee's endemism is also thought to contribute to its persistence in urban areas. In this study, only 9% of the bees found were considered exotic to North America (Packer et al., 2007). These results are consistent with other studies of exotic bees in urban environments. For example, a Canadian study found similar results of 14% (of 157 total bee species) and 11% (of 122 total bee species) of exotic bee species in Montreal and Quebec City, respectively. Another study of bees in New York City gardens reported that 19% of 54 total bee species were exotic (Matteson et al., 2008). Research in Michigan, USA, found that urbanization increased bee richness/abundance of exotic bees because they possessed traits that allowed them to survive better in urban areas (Fitch et al., 2019). As Ottawa's urbanization intensifies, exotic species may become more prevalent if they possess traits linked to urban survival.

Previous studies have found a lower proportion of ground nesting bees in cities (Cane et al., 2006; Geslin et al., 2016). Cities are thought to have a lack of bare ground, increased mulching, and irrigation which discourages ground nesting bee establishment (MacIvor, 2016). On the other hand, cavity nesters have novel nesting opportunities through man-made structures (Cane et al., 2006). Fifty percent of the bee species I caught were ground nesting. Since ~70% of eastern Canadian bee species nest in the ground (Packer et al., 2007), this is lower than expected, suggesting that ground nesters may be struggling in urban environments.

Cleptoparasitic bees are a group of bees that forgo nest construction and pollen collection and instead lay their eggs in the nests of other bee species (Michener, 2007). This lifestyle trait makes them a good indicator of bee populations as their presence indicates the existence of suitable hosts (Sheffield et al., 2013). I found that cleptoparasitic bees represented 15% of all bee species captured. The majority of these bees were in the generas *Nomada* and

Sphecodes, which is not surprising as they are parasites of *Andrena, Halictus,* and *Lasioglossum,* which were highly abundant in this study. These results are similar to other studies in urban environments. For example, a Springfield, MA study looking at bee composition in urban lawns, found 12.6% of their 111 collected bee species were cleptoparasitic (Lerman & Milam, 2016). Another study looking at suburban yards in Westchester County, NY found 19% of collected species were cleptoparasitic out of a total of 110 species (Fetridge et al., 2008). Although cleptoparasites generally have low populations, urban areas can support a large diversity of bee species (Hall et al., 2017; Matteson et al., 2008; Wojcik et al., 2008), which may also increase cleptoparasitic populations.

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Table E1. Ecological traits of collected bees

Species	Family	Abundance[1]	Origin[2]	Floral Preference[3]	Nesting[4]	Sociality[5]	Body Size[6]
Andrena (Taeniandrena) wilkella Kirby, 1802	Andrenidae	14	E	P	S	S	M
Andrena (Leucandrena) barbilabris Kirby, 1802	Andrenidae	1	N	P	S	S	-
Andrena (Melandrena) carlini Cockerell, 1901	Andrenidae	3	N	P	S	S	L
Andrena (Plastandrena) crataegi Robertson, 1893	Andrenidae	4	N	P	S	S	L
Andrena (Holandrena) cressonii Robertson, 1891	Andrenidae	1	N	P	S	S	-
Andrena (Melandrena) dunningi Cockerell, 1898	Andrenidae	16	N	P	S	S	L
Andrena (Andrena) mandibularis Robertson, 1892	Andrenidae	1	N	P	S	S	M
Andrena (Larandrena) miserabilis Cresson, 1872	Andrenidae	4	N	P	S	S	M
Andrena (Scrapteropsis) morrisonella Viereck, 1917	Andrenidae	3	N	P	S	S	M
Andrena (Simandrena) nasonii Robertson, 1895	Andrenidae	18	N	P	S	S	M
Andrena (Cnemidandrena) nubecula Smith, 1853	Andrenidae	2	N	O	S	S	-
Andrena(Melandrena) vicina Smith, 1853	Andrenidae	6	N	P	S	S	L
Andrena (Thysandrena) w-scripta Viereck, 1904	Andrenidae	5	N	P	S	S	M
Andrena (Simandrena) wheeleri Graenicher, 1904	Andrenidae	3	N	P	S	S	M
Calliopsis (Calliopsis) andreniformis Smith, 1853	Andrenidae	3	N	P	S	S	-
Pseudopanurgus rudbeckiae Robertson, 1895	Andrenidae	5	N	O	S	S	M
Apis (Apis) mellifera Linnaeus, 1758	Apidae	30	E	P	C	E	L
Bombus (Cullumanobombus) griseocollis De Geer, 1773	Apidae	10	N	P	C	E	L
Bombus (Cullumanobombus) rufocinctus Cresson, 1863	Apidae	7	N	P	C	E	L
Bombus (Pyrobombus) bimaculatus Cresson, 1863	Apidae	22	N	P	C	E	L
Bombus (Pyrobombus) impatiens Cresson, 1863	Apidae	40	N	P	C	E	L
Bombus (Pyrobombus) ternarius Say, 1837	Apidae	2	N	P	C	E	-
Bombus (Pyrobombus) vagans Smith, 1854	Apidae	12	N	P	C	E	L
Ceratina (Zadontomerus) calcarata Robertson, 1900	Apidae	53	N	P	P	B	S
Ceratina (Zadontomerus) dupla Say, 1837	Apidae	35	N	P	P	B	S
Ceratina (Zadontomerus) mikmaqi Rehan & Sheffield, 2011	Apidae	3	N	P	P	B	S
Epeolus pusillus Cresson, 1864	Apidae	3	N	P	[S]	S	M
Epeolus scutellaris Cockerell, 1824	Apidae	1	N	P	[S]	S	M
Melissodes (Eumelissodes) agilis Cresson, 1878	Apidae	5	N	O	S	S	L
Melissodes (Eumelissodes) trinodis Robertson, 1901	Apidae	13	N	O	S	S	-
Nomada articulata Smith, 1854	Apidae	17	N	P	[S]	S	M
Nomada bethunei Cockerell, 1903	Apidae	13	N	P	[S]	S	M

Species	Family	Abundance[1]	Origin[2]	Floral Preference[3]	Nesting[4]	Sociality[5]	Body Size[6]
Nomada cressoni Robertson, 1893	Apidae	7	N	P	[S]	S	M
Nomada denticulata Robertson, 1902	Apidae	1	N	P	[S]	S	M
Nomada depressa Cresson, 1863	Apidae	4	N	P	[S]	S	M
Nomada gracilis Cresson, 1863	Apidae	1	N	P	[S]	S	-
Nomada parva Robertson, 1900	Apidae	1	N	P	[S]	S	S
Peponapis (Peponapis) pruinosa Say, 1837	Apidae	8	N	O	S	S	L
Colletes compactus Cresson, 1868	Colletidae	2	N	O	S	S	-
Colletes nudus Robertson, 1898	Colletidae	1	N	P	S	S	L
Hylaeus (Hylaeus) annulatus Linnaeus, 1758	Colletidae	6	N	P	C	S	S
Hylaeus (Hylaeus) leptocephalus Morawitz, 1871	Colletidae	4	E	P	C	S	-
Hylaeus (Hylaeus) messilae Cockerell, 1896	Colletidae	4	N	P	C	S	S
Hylaeus (Prosopis) modestus/affinis Say/Smith, 1837/1853	Colletidae	24	N	P	C	S	S
Hylaeus undetermined	Colletidae	1	N	P	C	S	S
Agapostemon (Agapostemon) sericeus Forster, 1771	Halictidae	14	N	P	S	S	L
Agapostemon (Agapostemon) virescens Fabricius, 1775	Halictidae	34	N	P	S	S	L
Augochlora (Augochlora) pura Say, 1837	Halictidae	13	N	P	W	S	M
Augochloropsis (Paraugochloropsis) metallica Fabricius, 1793	Halictidae	1	N	P	S	B	M
Halictus (Odontalictus) ligatus Say, 1837	Halictidae	19	N	P	S	E	M
Halictus (Protohalictus) rubicundus Christ, 1791	Halictidae	25	N	P	S	E	L
Halictus (Seladonia) confusus Smith, 1853	Halictidae	56	N	P	S	E	M
Lasioglossum (Dialictus) species1 Curtis, 1833	Halictidae	12	N	P	S	-	S
Lasioglossum (Dialictus) species2 Curtis, 1833	Halictidae	2	N	P	S	-	M
Lasioglossum (Dialictus) species3 Curtis, 1833	Halictidae	10	N	P	S	-	M
Lasioglossum (Dialictus) species4 Curtis, 1833	Halictidae	4	N	P	S	-	M
Lasioglossum (Dialictus) species5 Curtis, 1833	Halictidae	1	N	P	S	-	M
Lasioglossum (Dialictus) species6 Curtis, 1833	Halictidae	2	N	P	S	-	S
Lasioglossum (Dialictus) species7 Curtis, 1833	Halictidae	33	N	P	S	-	S
Lasioglossum (Dialictus) species8 Curtis, 1833	Halictidae	3	N	P	S	-	S
Lasioglossum (Dialictus) species9 Curtis, 1833	Halictidae	18	N	P	S	-	S
Lasioglossum (Dialictus) species10 Curtis, 1833	Halictidae	1	N	P	S	-	S
Lasioglossum (Dialictus) species11 Curtis, 1833	Halictidae	1	N	P	S	-	S
Lasioglossum (Dialictus) species12 Curtis, 1833	Halictidae	6	N	P	S	-	S
Lasioglossum (Dialictus) species13 Curtis, 1833	Halictidae	7	N	P	S	-	S
Lasioglossum (Dialictus) species14 Curtis, 1833	Halictidae	33	N	P	S	-	S
Lasioglossum (Dialictus) species15 Curtis, 1833	Halictidae	18	N	P	S	-	S
Lasioglossum (Evylaeus) species16 Curtis, 1833	Halictidae	6	N	P	S	-	-
Lasioglossum (Evylaeus) species17 Curtis, 1833	Halictidae	1	N	P	S	-	-
Lasioglossum (Hemihalictus) species18 Curtis, 1833	Halictidae	16	N	P	S	-	-

Species	Family	Abundance[1]	Origin[2]	Floral Preference[3]	Nesting[4]	Sociality[5]	Body Size[6]
Lasioglossum (Leuchalictus) species19 Curtis, 1833	Halictidae	6	N	P	S	-	L
Lasioglossum (Leuchalictus) species20 Curtis, 1833	Halictidae	2	N	P	S	-	L
Lasioglossum (Leuchalictus) species21 Curtis, 1833	Halictidae	2	N	P	S	-	L
Sphecodes species1 Latreille, 1804	Halictidae	1	N	P	[S]	S	M
Sphecodes species2 Latreille, 1804	Halictidae	1	N	P	[S]	S	M
Sphecodes species3 Latreille, 1804	Halictidae	1	N	P	[S]	S	M
Sphecodes species4 Latreille, 1804	Halictidae	3	N	P	[S]	S	S
Sphecodes species5 Latreille, 1804	Halictidae	1	N	P	[S]	S	S
Anthidium (Anthidium) manicatum Linnaeus, 1758	Megachilidae	14	E	P	C	S	L
Anthidium (Proanthidium) oblongatum Illiger, 1806	Megachilidae	16	E	P	C	S	L
Chelostoma (Foveosmia) campanularum Kirby, 1802	Megachilidae	54	E	O	C	S	S
Chelostoma (Gyrodromella) rapunculi Lepeletier, 1841	Megachilidae	29	E	O	C	S	S
Coelioxys (Boreocoelioxys) rufitarsis Smith, 1854	Megachilidae	2	N	P	[C]	S	L
Heriades (Neotrypetes) carinata Cresson, 1864	Megachilidae	4	N	P	C	S	-
Heriades (Neotrypetes) leavitti Crawford, 1913	Megachilidae	7	N	P	C	S	S
Hoplitis (Alcidamea) producta Cresson, 1864	Megachilidae	14	N	P	P	S	M
Hoplitis (Alcidamea) spoliata Provancher, 1888	Megachilidae	1	N	P	C	S	L
Megachile (Chelostomoides) campanulae Robertson, 1903	Megachilidae	12	N	O	C	S	L
Megachile (Eutricharaea) rotundata Fabricius, 1787	Megachilidae	9	E	P	C	S	L
Megachile (Litomegachile) lippiae Cockerell, 1900	Megachilidae	1	N	P	S	S	-
Megachile (Megachile) centuncularis Linnaeus, 1758	Megachilidae	1	N	P	C	S	L
Megachile (Megachile) inermis Provancher, 1888	Megachilidae	2	N	P	C	S	L
Megachile (Sayapis) pugnata Say, 1837	Megachilidae	1	N	P	C/W	S	-
Megachile (Xanthosarus) frigida Smith, 1853	Megachilidae	2	N	P	C/W	S	L
Megachile (Xanthosarus) latimanus Say, 1823	Megachilidae	2	N	P	C	S	L
Osmia (Helicosmia) caerulescens Linnaeus, 1758	Megachilidae	4	E	P	C	S	L
Osmia (Melanosmia) albiventris Cresson, 1864	Megachilidae	1	N	P	C	S	-
Osmia (Melanosmia) pumila Cresson, 1864	Megachilidae	10	N	P	C/P	S	M

[1] Abundance is the total number of bees collected

[2] Origin: Each species is classified as native (N) or exotic (E) to North America

[3] Floral Preference: Species have been classified as polylectic (P), when pollen is collected from multiple flower families or oligolectic (O) when pollen is collected by a single family

[4] Nesting substrates were categorized as soil (S), cavity (C), pithy stem (P), and soft wood (W). Parasitic species were marked using [ ] around the category of their host. Species that have been documented using multiple substrates were marked with a " / "

[5] Sociality: was determined to be solitary (S), eusocial (E) or sub-social (B). Morphospecies are denoted with a "-" when sociality could not be determined

[6] Body size was determined based of the intertegular (ITD) distance average of five females. Large (L), medium (M), small (S). In species where Only males were collected ITD was not take (-)

(_) Underlined species had traits that were inferred from a close congener

Appendix F: Site Nesting Characteristics

Each site was assessed for the level of nesting resources. During each site visit, nesting resources were measured using a 1 m^2 quadrat. A total of four quadrats were placed during each site visit; two on the lawn and two in the garden. Quadrats were randomly placed where floral resources occurred and were at least 2m apart from each other. In cases where yards did not have any grass, four quadrats were placed in the garden instead. The area within the quadrat was assessed based on parameters determined to be important to ground nesting and cavity nesting bees (Potts et al., 2005; Potts & Willmer, 1997). These included percent bare soil, number of pithy stems, number of large cavities, and number of insect holes. The flowers within the quadrat were also counted as part of the indices (see the *Local Factors* section in the methods).

Sites were also assessed on the total nesting characteristics of the yard during the second sampling event. The characteristics were chosen based on Simao et al. (2018) and were measured based on their presence or absence within the yard (Table F1). Characteristics measured included: bare soil exposed to light for most of the day, dead wood, rock mulch (rock aggregations had to be greater than 1 cm), slope (presence or absence of an incline), large cavities, pithy stems, and if 50% of the pithy stems had established nests. Nesting characteristics were not included in my models to avoid overfitting.

Table F1. Site nesting characteristics

Site	% Garden[1]	% Grass[1]	% Impervious[1]	Bare Soil[2]	Dead Wood[2]	Slope[2]	Rock Mulch[2]	Large Cavities[2]	Pithy Stems[2]	Stems Nested[2]
S1	15	75	10	1	0	0	1	1	0	0
S2	50	40	10	1	1	1	0	0	1	1
S3	10	90	0	0	0	1	0	0	0	0
S4	10	40	50	1	0	0	1	0	1	0
S5	80	5	15	1	0	1	1	0	0	0
S6	20	60	20	1	0	1	0	0	1	1
S7	70	10	20	1	0	0	0	0	1	1
S8	35	40	25	1	0	1	1	0	0	0
S9	20	70	10	1	0	1	0	0	1	0
S10	50	45	10	1	0	1	0	0	1	1
S11	10	85	5	1	1	0	0	1	0	0
S12	60	0	40	1	1	1	0	0	1	1
S13	60	20	20	1	1	0	1	0	0	0
S14	45	15	40	1	0	1	1	0	1	0
S15	90	0	10	1	1	1	1	0	1	1
S16	35	55	10	1	1	0	1	1	1	1

Site	% Garden[1]	% Grass[1]	% Impervious[1]	Bare Soil[2]	Dead Wood[2]	Slope[2]	Rock Mulch[2]	Large Cavities[2]	Pithy Stems[2]	Stems Nested[2]
S17	75	20	5	1	1	1	1	1	1	1
S18	20	55	25	1	0	0	0	0	0	0
S19	80	0	20	1	1	1	0	0	1	1
S20	10	87	3	1	0	1	1	0	0	0
S21	20	10	70	1	0	0	0	0	1	1
S22	50	45	5	1	1	1	1	1	0	0
S23	50	35	15	1	1	1	1	0	1	1
S24	20	80	0	1	1	1	0	0	1	0
S25	35	60	5	1	0	0	0	0	1	1
S26	5	85	10	1	1	1	0	1	0	0
S27	50	0	50	0	1	0	0	0	0	0

[1] Percentage components were based on the yard as a whole

[2] Nesting variables were assessed on a presence (1) or absence (0) basis at each site

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