POLLINATOR CONSERVATION AND ACCESS TO POLLINATORS ON AN URBAN-RURAL GRADIENT

BY

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DISSERTATION

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ABSTRACT

Pollinators support global food security and provide important opportunities for people to interact with nature. These benefits are especially valuable in urban areas where city residents often rely on gardening as a supplemental source of food and income and where access to biodiversity and green space is often limited. Unfortunately, these benefits are threatened due to global declines in pollinators as well as inequities in access to urban biodiversity. This dissertation explores strategies that can be used to support pollinator conservation in urban areas and to increase city dwellers' access to the benefits provided by pollinators. My results reveal that access to pollinators along an urban-rural gradient in Illinois, USA may be limited by racial inequities. Fewer pollinators were observed in neighborhoods where more people of color live than in predominantly white neighborhoods. Additionally, I identify barriers to participation in pollinator-focused environmental education and volunteering that further limit opportunities to interact with pollinators. However, my results also suggest that conservation and management strategies that increase the availability of floral resources may also increase the abundance of pollinators along an urban-rural gradient. Bee abundance was found to be higher in registered pollinator gardens than in other types of urban green spaces. By implementing pollinator conservation strategies such as these, focusing on the accessibility of environmental education and volunteering, and addressing factors like systemic racism that lead to environmental inequities, we may be able to increase access to the many benefits provided by pollinators.

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CHAPTER 1: GENERAL INTRODUCTION AND LITERATURE REVIEW

Pollinators play an essential role in maintaining both ecosystem functioning and human well-being. Pollinators support the reproduction of approximately 85% of the world's plant species and many of these species would be fully incapable of reproduction without animal pollinators (Ollerton et al., 2011). Included in this figure are 87 of 115 leading global food crops. While many cereals such as corn, wheat, and rice are wind-pollinated, most fruits and vegetables, which provide micronutrients essential for human nutrition, rely at least in part on insect pollinators (Klein et al., 2007).

Additionally, pollinators provide important cultural ecosystem services. Due to their charismatic nature and the fact that they are relatively abundant and easy to observe, pollinators provide an excellent opportunity for people to interact with nature. Interaction with nature can have important positive impacts on human well-being (Miller & Hobbs, 2002; Miller, 2005). For example, people living near green spaces are physically and mentally healthier on average than those who live farther away (van den Berg et al., 2015). Additionally, interaction with nature has positive developmental outcomes and can provide valuable educational opportunities for both children and adults. Spending time in green spaces provides children with opportunities to make observations and practice critical thinking, feel connected to and engaged with their surroundings, and develop a personal sense of identity (Kellert, 2002). In recent years, a number of environmental education programs focusing on pollinators have been developed and implemented with success (Saunders et al., 2018).

Unfortunately, these ecosystem services are currently threatened by global declines in many species of pollinators. Honey bee populations are declining in both Europe and North

America, with the number of honey bee colonies in the United States having decreased by 55% since 1947 (NASS, 2018; Potts et al., 2010b; van Engelsdorp et al., 2008). While less research has been conducted focusing on communities of native bees, similar negative trends have been documented (Cameron et al., 2011; Goulson et al., 2008; Potts et al., 2010a). For example, a study using historical records in central Illinois found that over half of wild bee species observed in the late 1800s were extirpated by 2010 (Burkle et al., 2013). In Europe, overall butterfly abundance and richness is in decline, and extinction rates are high (Bartomeus et al., 2018; Warren et al., 2021). In North America, trends in butterfly populations appear to be associated with climate change. Declines in overall butterfly abundance occurred in regions that are becoming hotter and drier, while an increase in abundance occurred in regions that are becoming cooler and wetter (Crossley et al., 2021). While pollinating flies seem to be faring better and their richness is even increasing in some parts of the world (Goulson et al., 2015; Sanchez-Bayo & Goka, 2014; Thogmartin et al., 2017), declines have also been observed for some species and lack of data prevents a clear understanding of global population trends (Bartomeus et al., 2018; Beismeijer et al., 2006; Carvalheiro et al., 2013).

While the causes of declines in pollinator populations are not fully understood, many interacting factors are thought to contribute including exposure to pesticides and herbicides, introduction of new parasites and diseases, and habitat loss and fragmentation (Goulson et al., 2015; Potts et al., 2010a). Pesticides including neonicotinoids and organophosphates have both lethal and sublethal effects on native bees, managed honey bees, and butterflies, causing reduced activity levels and lower body weights (Goulson et al., 2015; Sanchez-Bayo & Goka, 2014; Thogmartin et al., 2017). Additionally, herbicides such as glyphosate negatively impact pollinators by removing floral resources and host plants. This has been particularly problematic for species such as the monarch butterfly that rely on an obligate host plant (Thogmartin et al., 2017). Parasites and diseases such as *Varroa destructor*, *Crithidia bombi*, and *Aeithina tumida* that have been introduced as a result of the transportation of domestic bees have had disastrous consequences for wild bumble bees in many parts of the world, sometimes leading to the collapse of entire populations (Goulson et al., 2015). Finally, anthropogenic habitat loss has been identified as one of the most important drivers of reductions in pollinator abundance and diversity (Goulson et al., 2005; Hung et al., 2017; Winfree et al., 2009). Habitat destruction and degradation leads to a reduced availability of floral resources used by adult pollinators as well as suitable nesting sites for bees and host plants used by butterflies (Lazaro & Tur, 2017; Roulston & Goodell, 2011; Thogmartin et al., 2017).

Understanding how to address these threats and support pollinator conservation is of particular importance in urban areas. While several studies have indicated that urbanization is associated with a reduced abundance and diversity of pollinators (Cardoso & Gonçalves, 2018; Fontaine et al., 2016; Glaum et al., 2017; Hernandez et al., 2009), there is also evidence emerging that urban areas may provide important resources for many species (Hall et al., 2017; Lynch et al., 2021; Samuelson et al., 2021). For example, urbanization can lead to an increased abundance of floral resources as well as a longer flowering period (Harrison et al., 2018a; Lynch et al., 2021). This may be particularly true in agricultural landscapes where the rural end of the urbanization gradient is dominated by mass flowering crops that bloom for a relatively short period of time (Galpern et al., 2017; Hall et al., 2017; Shaw et al., 2020). Bees in particular appear to respond well to urbanization and may have increased abundance and diversity in urban areas (Baldock et al., 2015; Sirohi et al., 2015; Winfree et al., 2007). Additionally, one recent study found that honey bees in cities travel shorter distances in search of food than honey bees in rural areas (Samuelson et al., 2021).

Understanding the potential of urban areas for pollinator conservation will help us develop management strategies that can be used to maximize their conservation value. Pollinator conservation has gained increased attention in recent years as scientists, managers, and the public have become aware of the potential impacts of declining pollinator populations, and public action to improve pollinator habitat has been particularly strong in cities (Byrne & Fitzpatrick, 2009; Vilsack & McCarthy, 2016). Initiatives such as pollinator garden registration programs that encourage community members to provide resources such as pollen and nectar, larval host plants for butterflies, and structures that can be used for nesting by bees have become common, and interest in these types of programs is increasing (Majewska & Altizer, 2020).

Despite the increasing public involvement in urban pollinator conservation initiatives, little research has been done to evaluate their effectiveness or to develop an evidence-based approach to designing urban pollinator habitat. However, research identifying habitat features that benefit pollinators in urban areas can help us begin to identify effective conservation strategies. One recent review found that pollinators in urban gardens benefit from high plant species diversity, the presence of woody vegetation, large garden size, and sun exposure (Majewska & Altizer, 2020). Additionally, evidence suggesting that pollinators are responding positively to management interventions is beginning to emerge. For example, a national study evaluating changes in butterfly populations found that overall trends in the abundance of butterflies were more positive in urban than in rural areas, suggesting that butterfly-friendly practices implemented in cities may be having a positive effect (Crossley et al., 2021).

The high potential of cities for pollinator conservation may be particularly valuable because these are also areas where the benefits that pollinators provide are especially needed. Pollination services support food security in urban areas where city residents often rely on gardening as a supplemental source of food and income and may help to combat the impacts of urban food deserts (Abdulkadir et al., 2012). Additionally, access to biodiversity and green space is often limited in cities, making interaction with charismatic and easy to observe pollinators especially beneficial. Pollinator-focused conservation and research provide meaningful opportunities for interaction with local ecosystems that are relatively accessible to city-dwellers. For example, the multitude of pollinator-focused community science projects that have been developed and implemented in recent years may be especially beneficial to people living in cities (Koffler et al., 2021; Ries & Oberhauser, 2015). Community science refers to the participation of people who are not professional scientists in academic research and is known to increase interest and self-confidence regarding science and facilitate the development of a strong connection to place (Hiller & Kitsantas, 2014; Toomey et al., 2020). The terms “community science” and “citizen science” are often used interchangeably, and we have opted to use the term “community science” in this dissertation to clarify that citizenship of the United States or any other country was not required for participation in the project.

City-dwellers will only receive the full benefits provided by pollinators if we can ensure that all urban residents are able to access pollinators and environmental education opportunities that connect people to nature. In addition to challenges presented by global pollinator declines, inequities in access to nature create a barrier that prevents many urban residents from interacting with pollinators. It is well known that people living in lower income urban neighborhoods often have less access to green space and biodiversity than people living in wealthier neighborhoods (Hope et al., 2003a; Leong et al., 2018; Schell et al., 2020). The positive correlation between wealth and biodiversity is referred to as the "luxury effect" and was first identified in 2003 by ecologists working in Phoenix, Arizona (Hope et al., 2003a). This pattern is thought to occur because wealthier people are able to spend more money landscaping and because local governments often direct more funding towards green space management in wealthier neighborhoods (Schell et al., 2020; Wolch et al., 2005). While the luxury effect appears to have particularly strong impacts on plant communities (Leong et al., 2018), it has also been detected in studies focusing on birds (Kinzig et al., 2005; Lerman & Warren, 2011), mammals (Magle et al., 2016), reptiles (Ackley et al., 2015), and arthropods (Leong et al., 2016). Since green space in cities tends to be planted and carefully managed by people, the luxury effect is expected to be strongest in urban areas, and indeed, most evidence of this pattern comes from studies conducted in cities (Leong et al., 2018). However, there is some evidence that this pattern may also occur in rural areas, and more research is needed to understand how relationships between wealth and biodiversity may vary along an urban-rural gradient (Chamberlain et al., 2019; Li et al., 2019).

Inequities based on race also shape access to nature in urban areas. There is often less green space and lower biodiversity in neighborhoods where people of color live than in predominantly white neighborhoods (Lerman & Warren, 2011; Saporito & Casey, 2015). Tree canopy tends to be lower in these neighborhoods and the urban heat island effect is stronger

(Flocks et al., 2011; Jesdale et al., 2013). These trends likely reflect discriminatory practices that direct funding away from neighborhoods where people of color live. For example, the lagging impacts of historic housing discrimination practices may continue to shape the ecology of cities (Schell et al., 2020). Between 1935 and 1968, the federal government implemented a policy that is often referred to as "redlining." This policy instructed the Home Owner's Loan Corporation to develop maps that categorized urban neighborhoods into four grades. Neighborhoods given a classification of A were considered to be the safest areas for investment and people living in those neighborhoods were able to easily access loans to buy or refinance homes, while neighborhoods given a classification of D were considered to be "hazardous" and people in those areas were rarely able to access housing loans. In many cases, the classification of neighborhoods was based explicitly on race (Rothstein, 2017). In neighborhoods assigned a grade of D, landlords who were unable to access loans often abandoned their properties leading to reduced property tax income for the neighborhood. As a result, these neighborhoods had less funding available to spend on public services including conservation and the maintenance of green space (Schell et al., 2020).

Although the practice of redlining has been illegal for more than 50 years, evidence suggests that tree canopy cover and access to green space continues to be lower in historically redlined neighborhoods (Locke et al., 2021; Namin et al., 2020; Nardone et al., 2021). While we are beginning to understand some of the relationships between race, discrimination, and urban land cover, ecologists are calling for more research focusing explicitly on the impacts of discriminatory practices on the ecological functioning of cities (Schell et al., 2020). A clearer understanding of the mechanisms leading to reduced access to green space and biodiversity in neighborhoods where people of color live may help us to correct inequities and support effective biodiversity conservation in neighborhoods that have been impacted.

More research focusing on the impacts of social inequities on urban pollinators is especially needed given the essential ecosystem services that they provide and the current lack of information available. The few studies focusing on the impacts of the luxury effect on pollinator communities have had mixed results. Overall pollinator abundance in UK urban gardens was higher in neighborhoods with a higher median household income (Baldock et al., 2019), and the abundance and diversity of bees collected from nest boxes in Toronto, Canada was higher in wealthier neighborhoods (MacIvor, 2015). However, another study found no impact of the luxury effect on butterflies in Israeli cities (Keren et al., 2022). To my knowledge, no studies have yet been published focusing on the impacts of race or discriminatory practices on urban pollinators.

Limited accessibility of environmental education and community science programs that provide opportunities for people connect with nature may also prevent urban residents from receiving benefits provided by pollinators. Environmental education often perpetuates colonial ideas such as the concept of wilderness, erases indigenous histories, and occurs in white-dominated spaces where people of color frequently experience exclusion and microaggressions (McLean, 2013; Romero et al., 2022; Warren et al., 2014). As a result, diversity within the field remains low despite efforts to promote equity and inclusion (Romero et al., 2022). Similar patterns are observed in community science, with people belonging to groups that are historically underrepresented in science being less likely to participate (Hobbs & White, 2012; Pandya, 2012).

The COVID-19 pandemic has created additional barriers to participation in environmental education and community science. Many of these programs paused activities as a result of social distancing requirements and most programs that continued to operate transitioned to an online format (Dwivedi, 2020; Escoto-Murillo & Alfaro, 2021; Lepenies & Zakari, 2021; Smith & Hamed, 2020). As a result, computer and internet access were required for participation. Additionally, many people faced difficult life circumstances such as illness, death of loved ones, and a lack of access to childcare that made participation in these types of programs more difficult. These barriers were particularly problematic for people with limited access to technology or who continued to work in-person during the pandemic (Smith & Hamed, 2020; Van Haeften et al., 2020). Understanding the barriers that prevent people from participating in nature-focused education and volunteering can help us create more opportunities for people to interact with local ecosystems and ensure that all urban residents are able to access the benefits provided by pollinators and other urban wildlife.

This dissertation explores strategies that can be used to support pollinator conservation in urban areas and to increase city dwellers' access to the benefits provided by pollinators. The first chapter evaluates whether pollinator garden registration programs are an effective tool for bee conservation along an urban-rural gradient. The second chapter explores equity of access to pollinator biodiversity by examining the relationships between income, race, and discriminatory practices and pollinator visitation rates. Finally, the third chapter considers the impacts of the COVID-19 pandemic on city dwellers' opportunities to interact with and learn from pollinators. I describe an example of a pollinator-focused community science project that I implemented during the first summer of the pandemic and explore strategies that could be used to address barriers to participation in community science both in the context of the pandemic and moving forward into a post-pandemic world.

CHAPTER 2: EFFECTIVENESS OF POLLINATOR GARDENS FOR BEE CONSERVATION ALONG AN URBAN-RURAL GRADIENT[1]

Abstract

With growing awareness of global declines in bee populations, bee conservation has received increased attention from governmental agencies, non-profit organizations, researchers, and the public. One popular strategy involves pollinator garden registration programs that certify gardens meeting required habitat provisioning criteria to motivate gardeners to provide high-quality food and nesting resources. While considerable financial and human resources are directed towards these programs and there are now over one million pollinator gardens registered globally, limited research has been conducted to verify their effectiveness for bee conservation. We compared bee visitation to pollinator gardens, ornamental gardens, and lawns on an urban-rural gradient in the Greater Chicago Area, IL USA. Our results suggest that pollinator gardens may support bee conservation by providing important floral resources. We observed more bees visiting pollinator gardens than ornamental gardens or lawns across the full urban-rural gradient, with this pattern being most pronounced for honey bees. Based on these results, we suggest that a continued emphasis on pollinator garden registration programs may be an effective strategy for bee conservation but recommend an increased focus on native bees.

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[1] L.R. Lynch, and J.R. Miller. Prepared for submission to *Urban Ecosystems*

Introduction

Approximately 85% of the world's plant species are pollinated by animals (Ollerton et al., 2011), and 35% of global food production depends at least partially on insect pollination (Klein et al., 2007). Bees' pollen-carrying structures make them one of the most efficient groups of pollinators (Batra, 1995). In recent years, pollination services have been threatened due to global declines in bee populations. The number of honey bee colonies in the United States has declined by 55% since 1947 (NASS, 2018; van Engelsdorp et al., 2008), and abundance and diversity of native bees have been declining in many parts of the world (Cameron et al., 2011; Goulson et al., 2015; Potts et al., 2010a).

Declines in bee populations are driven by a variety of interacting factors (Goulson et al., 2015; Potts et al., 2010a). Introduced parasites and diseases have had disastrous consequences (Goulson et al., 2015) and pesticides such as neonicotinoids and organophosphates may kill bees directly or, in smaller doses, reduce the ability of hives to survive the winter (Sanchez-Bayo & Goka, 2014). Habitat loss is often identified as one of the greatest threats to bee populations (Goulson et al., 2015; Potts et al., 2010a). One meta-analysis of the effects of anthropogenic disturbance on bees found habitat loss to be the only factor causing significant declines in native bee abundance and diversity (Winfree et al., 2009). Habitat loss causes declines mainly by reducing the availability of necessary resources such as nectar, pollen, and suitable nesting sites (Lazaro & Tur, 2017; Roulston & Goodell, 2011). Food availability is thought to be the most important direct driver of community composition, with bee abundance and diversity being positively correlated with flower abundance (Potts et al., 2003; Roulston &

Goodell, 2011; Williams et al., 2015) and with the presence of certain species of flowers that are particularly attractive to bees (Harmon-Threatt & Hendrix, 2015).

Due to bee population declines, conservation organizations are directing more of their time and funding toward bee conservation. (Byrne & Fitzpatrick, 2009; Vilsack & McCarthy, 2016). Much of this effort focuses on outreach to encourage community members to improve habitat through activities such as planting flowers that are attractive to bees and providing structures for nesting (Burkle et al., 2017; Penn et al., 2018). A number of organizations have developed pollinator garden registration programs that certify gardens meeting required habitat provisioning criteria. These gardens are expected to support bee conservation by providing high-quality floral resources that are available throughout the growing season. There are currently more than one million such gardens registered globally (National Pollinator Garden Network, 2021).

Despite the considerable resources directed towards these initiatives, limited research has been conducted to verify their effectiveness as a strategy for conservation. Management steps taken to increase floral resource availability can increase the abundance and diversity of bee communities (Shwartz et al., 2013; Williams et al., 2015) and one study found a diverse assemblage of bee species in a campus pollinator garden (Shivalingaswamy et al., 2020). However, to our knowledge, no study has evaluated the effectiveness of pollinator gardens by comparing bee communities observed in registered pollinator gardens to those observed in other types of green spaces.

In this study, we examine the attractiveness of registered pollinator gardens to bees on an urban-rural gradient in the Greater Chicago Area, IL USA. We predicted that the abundance and richness of flowering plants would be greater in pollinator gardens than in ornamental gardens or lawns, that bee abundance and richness would be positively associated with flower abundance and richness, and that bee abundance and richness would therefore also be highest in pollinator gardens. Because bee populations are driven by floral resource availability (Roulston & Goodell, 2011), human choices largely determine flower communities in gardens and lawns, and the gardening decisions that people make vary by neighborhood demographics (Lowenstein & Minor, 2016), we also tested for the possibility that the attractiveness of pollinator gardens could vary along an urban-rural gradient. Due to social pressures to maintain “neat” yards, people living in urban and suburban areas may manage their gardens and lawns more intensively, reducing biodiversity by cutting down on weedy plants (Grove et al., 2006). As a result, we predicted that pollinator gardens may be comparatively more attractive in cities and suburbs than in rural areas.

Methods

Study Area and Sampling Design

Study sites (n=39) were located in Cook, DuPage, Kane, Grundy, and Will Counties, IL USA (Fig 2.1). Rural land surrounding Chicago is composed largely of cropland interspersed with pasture, forest, and remnant prairie (Dewitz, 2019). Impervious surface cover within a 1-km radius of each study site ranged from 1% to 82% and the percent cover of agricultural land within a 1-km radius of each study site ranged from 0% to 95%.

At each site, a registered pollinator garden was matched with an ornamental garden and a lawn located within 1.5 km. This allowed us to compare bees’ use of local resources found in different types of green spaces while reducing the variation in landscape characteristics that may affect bee communities. Each set of three study sites was located at least five km from all other sets. In 2018, 10 sets of 3 sites were used. In 2019, three additional sets of sites were added. Site area ranged from 25 to 3000 m^2.

All pollinator gardens were registered through the Monarch Waystation Program (Landis, 2014). Gardeners who wish to register their garden as a Monarch Waystation can do so by certifying that their garden contains several species of flowering plants that bloom continuously throughout the growing season, contains at least 10 milkweed plants, receives at least 6 h/d of sun, and has a density between 2.2 and 10.9 plants/m^2. Native species are recommended but not required. Ornamental gardens comprise plots of flowering plants that were not registered as pollinator gardens. Lawns were regularly mowed areas in which vegetation consisted mainly of grasses, clover, and other herbaceous plants.

Bee Communities

To measure the ways in which bee community composition varied based on habitat characteristics, we sampled bees biweekly in June, July, and August of 2018 and 2019 using bee bowls and visual surveys. A combination of passive and active sampling allows for a more accurate characterization of bee communities (Grundel et al., 2011; Wilson et al., 2008). While bee bowls are most frequently used in conjunction with hand netting (Lerman & Milam, 2016; Matteson et al., 2008; Winfree et al., 2007), we replaced this technique with visual observations to prevent damage to ornamental plants (Matteson et al., 2013).

Two sets of three bowls (fluorescent yellow, fluorescent blue, and white) were placed at each study site to capture bees attracted to flowers of different colors (Droege, 2005; Wilson et al., 2008). To standardize their placement, sets of bowls were arranged 1 m from the edge of the study site, with one set in the northeastern corner of the site and the other in the southwestern corner. We elevated the bee bowls to the height of the surrounding vegetation (Glaum et al., 2017) and filled them with water and a drop of unscented dish detergent to break the surface tension. We left bee bowls at each site for 24 h (Lerman & Milam, 2016).

Study sites were monitored for 30 min, and all bees observed on the reproductive parts of a flower during this time were recorded. Observers walked at a pace that allowed them to sample the entire garden and spent more time observing the most active flowers. They began at a fixed point each time they sampled and then moved from flower to flower as they walked through the gardens (Hülsmann et al., 2015). Bees were classified as honey bees, bumble bees, or other bees. These categories were chosen because they are easily distinguishable in the field and align with categories that were used by community scientists participating in a concurrent study. Additionally, we identified all bumble bees that we observed to species using the guide *Bumble Bees of the Eastern United States* (Colla et al., 2011).

Sampling was carried out on sunny days with a temperature of $\geq$15.5 °C and wind speeds < 20 km/h (Harmon-Threatt & Hendrix, 2015). Data were collected by L. Lynch and five field technicians. Before beginning data collection, each technician conducted two practice surveys concurrently with L. Lynch to reduce variability.

Bees collected in bowls were identified to species using the guide *Bees of the Tallgrass Prairie Region and Greater Midwest* written by Mike Arduser (Arduser, 2020). Other guides were used for genuses that were not included in this guide. Males from the genus *Lasioglossum* were identified using, “Revision of the metallic species of *Lasioglossum* (*Dialictus*) of eastern

North America (Hymenoptera, Halictidae, Halictini)” and “Revision and reclassification of *Lasioglossum (Evylaeus*), *L*. (*Hemihalictus*) and *L*. (*Sphecodogastra*) in eastern North America (Hymenoptera: Apoidea: Halictidae)” (Gibbs, 2011; Gibbs et al., 2013). Females from the genus *Melissodes* and males from the genus *Megachile* were identified using keys available on the DiscoverLife website (Discover Life, 2022). Most specimens were identified by L. Lynch, but identifications were confirmed by Mike Arduser when there was uncertainty.

Floral Resources

To measure the availability of floral resources at each study site, we identified all flowering plants, recording the number with open blossoms as well as the number of open blossoms on each plant (Lerman & Milam, 2016; Matteson et al., 2013; Winfree et al., 2007). Based on this information, we calculated flower density and flower richness per m^2 at each study site for each observation period. We surveyed plants biweekly in June, July, and August on the same days that bee surveys were carried out. Temperature, wind, cloud cover, and observer were also recorded at each study site at each observation period.

Landscape Habitat Characteristics

To examine the relationships between landscape context, bee communities, and bee conservation efforts, we used impervious surface cover as a measure of urbanization in the landscape surrounding each site (Lu & Weng, 2006; McDonnell & Hahs, 2008). Impervious surface cover was measured within concentric circles with radii matching the typical daily traveling distance of the bee taxa that we considered. For honey bees and bumble bees, we used a 2-km radius because these species typically forage within 2 km of their hives (Carr-Markell et al., 2020; Zurbuchen et al., 2010). For overall bee abundance and richness, we used a

750-m radius because many of the species we expected to be common at our study sites are smaller in size and therefore have smaller foraging distances (Greenleaf et al., 2007). We measured impervious surface cover in ArcGIS (ESRI, 2011) using data from the National Land Cover Database (Dewitz, 2019). A State Plane Illinois East projection was used for all maps.

While we were most interested in considering urbanization as a continuous gradient (McDonnell & Pickett, 1990; McDonnell & Hahs, 2008), we also created a categorical variable to visualize interaction effects between urbanization and local resource availability. Study sites with < 20% impervious surface cover in the surrounding landscape were categorized as "rural," sites with 20%-50% impervious surface cover as "suburban," and sites with > 50% impervious surface cover as "urban" (Potere et al., 2009).

Analyses

We used linear mixed-effects models to examine the relationships between flower density and the abundance of honey bees, the abundance and richness of bumble bees, and total bee abundance and richness (Harrison et al., 2018b). For each dependent variable, we used AIC_C to compare four models (Anderson et al., 2000). The first model included only flower density; the second included the interaction effects between flower density and impervious surface cover; the third included the additive effects of flower density and impervious surface cover; and the final model was the null. The same models were used to examine the impacts of flower richness on bee abundance and diversity. All models included study site as a random variable to address nonindependence from repeated sampling. Additionally, we used AIC_C to identify variables to be included as covariates. Potential covariates included study site area, year, day, time, wind speed, cloud cover, temperature at time of sampling (for active sampling models), and daily high temperature (for passive sampling models). We compared models including the individual effects of each of these potential covariates and included covariates in the final set of models if they ranked above the null and their AIC_C weights added up to 0.95 (Burnham & Anderson, 2002). The dependent variable of each model was square-root transformed to improve the ability to meet assumptions of normality and constant variance.

Because increasing floral resource availability is a goal of pollinator garden registration programs, we used a separate AIC_C comparison to examine the relationships between site type (pollinator garden, ornamental garden, or lawn), urbanization, and bee abundance and richness. Site type and urbanization were included in these models as continuous variables. We used linear mixed-effects models to describe the relationships between site type, impervious surface cover and the abundance of honey bees, the abundance and richness of bumble bees, and total bee abundance and richness. For each dependent variable, we compared seven models. The first two included only site type and only impervious surface cover. The third included the additive effects of site type and impervious surface cover while the fourth included the interaction effects of these two variables. The fifth included the quadratic relationship between impervious surface cover and the dependent variable. The sixth and seventh included the additive and interaction effects, respectively, of impervious surface cover as a quadratic variable and site type. The final model was the null. A square root transformation was used for all models and random variables and covariates were selected in the same manner as described in the previous paragraph. Additionally, we used the same procedure and set of models described above to examine the relationships between site type, urbanization, and the density and richness of floral resources.

For all AIC_C comparisons, we considered models with $\Delta AIC_C < 2$ to be competitive (Burnham & Anderson, 2002). To avoid analyses based on uninformative parameters, we identified models for which a simpler nested model was also included in the set of competitive models. Parameters present in the more complex model but not the nested model were considered to be potentially uninformative. Additionally, parameters in models within two AIC_C points of the null model per additional parameter were considered potentially uninformative (Arnold, 2010). Models including potentially uninformative parameters were retained if the 85% confidence interval of at least one of these parameters did not overlap zero or if at least one of these parameters also appeared alone in another model that ranked above the null (Arnold, 2010). Otherwise, models were removed from the set of competitive models.

We graphed the predicted values of dependent variables based on the informative parameters present in the competitive model sets. Model-averaging was used to calculate predicted values when multiple models including the same parameter were present in the set of competitive models (Burnham & Anderson, 2002). Models used to visualize interaction effects included site type and urbanization as categorical variables. All analyses were conducted in R (R Core Team, 2021).

Results

Bee Community

We collected a total of 898 individuals using bee bowls, including 73 different species (Table 2.1). The three most common species were *Melissodes bimaculatus* (20.94% of individuals), *Agapostemon virescens*, (13.81% of individuals), and *Lasioglossum hitchensi* (9.47% of individual). We observed a total of 5,289 individuals through active sampling. This included 1,989 *Apis mellifera* individuals and 659 individuals from the genus *Bombus* belonging to nine species.

Floral Resources

We observed a positive relationship between flower density and the abundance and richness of bees visiting our study sites. Flower density was included as a variable in the competitive model set for all dependent variables considered (Table 2.2; Fig. 2.2). Models including interaction effects between flower density and urbanization were also competitive when considering the abundance and richness of bees collected in bee bowls. Bee abundance and richness increased strongly with flower density at rural sites but were not related to flower density at urban sites. A slight positive relationship was observed between flower density and bee richness at suburban sites (Fig. 2.3). See Appendix A for the complete AIC_C tables associated with the analyses for this chapter.

Similarly, we found a positive relationship between flower richness and the abundance and richness of bees visiting our study sites, with flower richness included as a variable in the competitive model set for all dependent variables considered (Table 2.3; Fig. 2.4). For honey bee abundance, bumble bee abundance, and bumble bee richness based on active sampling as well as total bee abundance based on passive sampling, models including the interaction effects between flower richness and urbanization were also competitive. The relationships between flower richness and honey bee abundance, bumble bee abundance, and bumble bee richness based on data from surveys were stronger at urban sites than at rural or suburban sites. The relationship between flower richness and the number of bees collected in bee bowls was strongest at rural study sites (Fig. 2.5).

Urbanization

The abundance of honey bees and the richness of bumble bees was found to increase linearly with urbanization (Table 2.4; Fig. 2.6). For bumble bee abundance, models including both linear and quadratic relationships with urbanization were competitive. Based on model-averaged values from competitive models, bumble bee abundance was found to peak at 54% impervious surface cover. No relationship was observed between urbanization and the overall abundance or richness of bees.

Pollinator Gardens

Flower density was highest in pollinator gardens and lowest in lawns while flower richness was about the same in pollinator gardens and ornamental gardens and lower in lawns (Fig. 2.7). The abundance and richness of all groups of bees considered was also highest in pollinator gardens (Fig. 2.8). This pattern was strongest for honey bees. Honey bee abundance was three times higher in pollinator gardens than in ornamental gardens while overall bee abundance was only about two times higher in pollinator gardens. The relationship with site type was weakest for overall bee richness, which was only about 1.7 times higher in pollinator gardens.

For bumble bee richness, the competitive model set also included the interaction effects between site type and urbanization (Fig. 2.9). The difference between bumble bee richness observed in pollinator gardens and ornamental gardens was smallest at rural sites and largest at urban sites.

Discussion

Our results suggest that pollinator gardens are attractive to bees. A greater abundance and richness of bees were observed in pollinator gardens than in ornamental gardens, likely due to the increased availability of floral resources. Flower density was higher in pollinator gardens than in ornamental gardens, and bee abundance and richness were positively associated with flower density. These results align with our predictions and with numerous studies indicating that floral resources are a key driver of bee abundance and richness (Potts et al., 2003; Roulston & Goodell, 2011; Williams et al., 2015).

The relationship between floral resource availability (flower density and richness) and the abundance and richness of bees collected in bee bowls was strongest in rural areas, whereas the relationship between floral resource availability and the abundance and richness of bees observed through surveys was strongest in urban areas. These differences may have to do with the fact that the effectiveness of sampling methods can vary depending on landscape context (Prendergast et al., 2020). It is well-known that the abundance, diversity, and species composition of bees sampled is related to the sampling methods used (Krahner et al., 2021). One recent study found that in urban areas, the highest abundance of bees was recorded when surveys were used while very low catch rates were observed in bee bowls (Prendergast et al., 2020).

While bee abundance was highest in pollinator gardens for all groups of bees, they appeared to be most attractive to honey bees. Honey bee abundance was more than three times higher in pollinator gardens than in ornamental gardens, while total bee visitation was only about twice as high. This difference may reflect the fact that these gardens provide resources that more closely match the needs of honey bees than native bee species. This would be unsurprising considering that scientific research as well as public outreach often focus heavily on honey bee conservation (Goulson et al., 2015; Smith et al., 2016). In contrast, scientific understanding of the status of native bee populations is much less complete (Goulson et al., 2015), and one study found that only 15% of stories regarding pollinators in the popular media mentioned native bees (Smith et al., 2016).

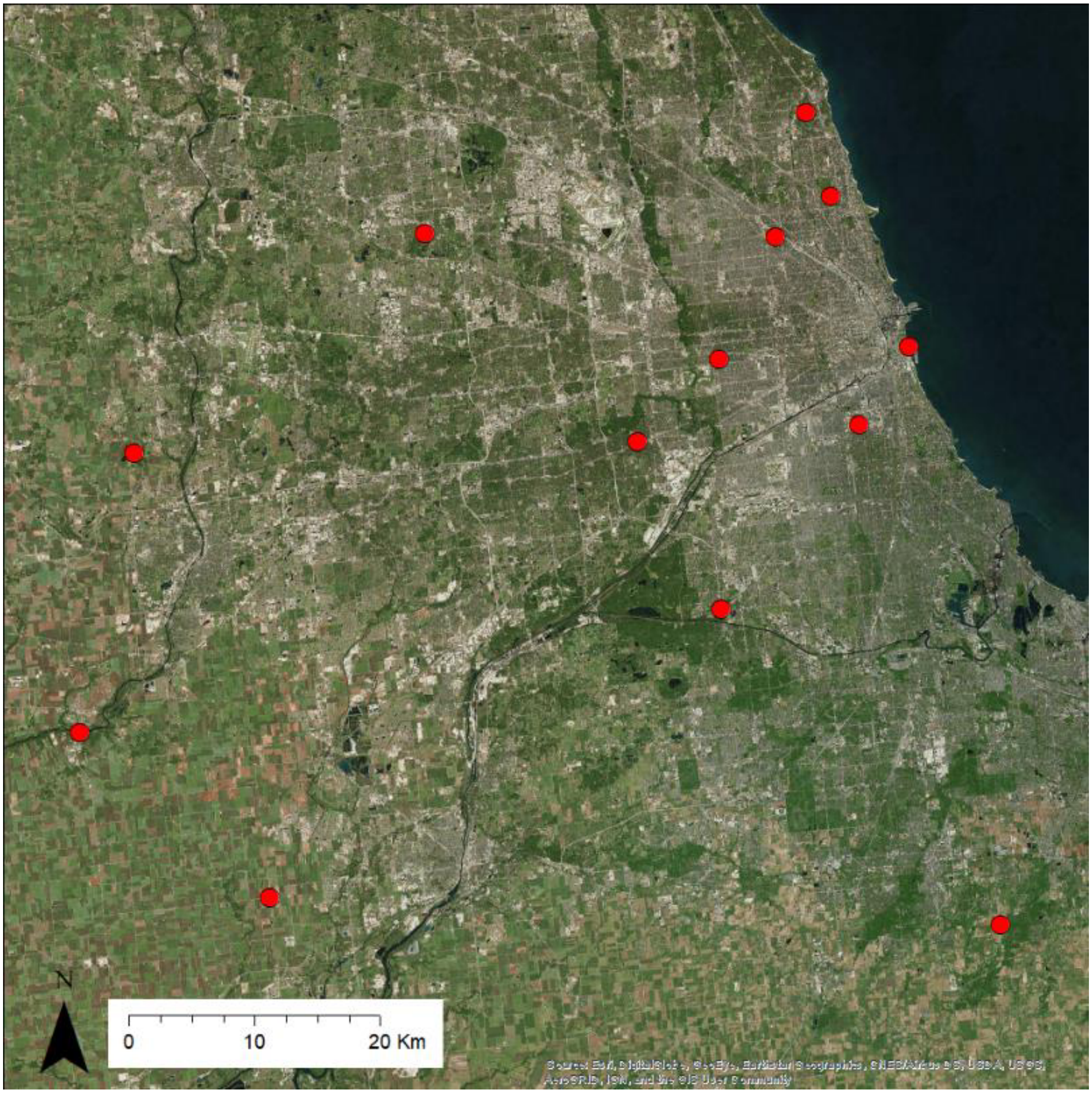
Our results suggest that the attractiveness of pollinator gardens to bumble bees may increase with urbanization. Given that the attractiveness of ornamental gardens and lawns does not appear to decrease along the urban-rural gradient, it may be that gardeners are implementing more effective strategies for bumble bee conservation in urban pollinator gardens. We did not find any evidence that the attractiveness of pollinator gardens to honey bees or overall bee communities varied with urbanization. It may be that other social factors play a more important role in determining the behavior of gardeners. For example, pressure to maintain a “neat” yard with fewer weedy flowers may be higher in wealthier neighborhoods (Robbins et al., 2001; Zhou et al., 2009).

We did observe relationships between urbanization and the abundance and richness of bees. Honey bee abundance and bumble bee richness increased linearly with urbanization while bumble bee abundance peaked at an intermediate level of urbanization. Although we did not observe changes in flower density or richness along an urbanization gradient at our study sites, these patterns may still be driven by floral resource availability at a landscape scale. Patterns of floral resource availability may vary depending on land use/land cover (LULC), and the distribution of LULC categories may also vary along an urban-rural gradient (Lynch et al

2021). For example, there may be a greater density of gardens in urban areas where houses are closer to each other. It has been suggested that urban and suburban areas may provide a ‘refuge’ for bees in landscapes in which rural areas are dominated by intensive agriculture (Hall et al., 2017), and several other studies have found bee abundance and diversity to increase with urbanization (Baldock et al., 2015; Sirohi et al., 2015; Winfree et al., 2007) or observed peaks in pollinator diversity at intermediate levels of urbanization (Fortel et al., 2014).

Based on our results, it appears that the Monarch Waystation program may be successfully increasing the availability of food resources and the abundance of bees. Efforts to expand participation in pollinator garden registration programs are therefore likely to benefit bees along an urban-rural gradient. However, we suggest that these programs could increase their conservation value further by providing gardeners with recommendations about how to improve habitat specifically for native bees as well as for honey bees.

Tables and Figures



Source: Esri, DigitalGlobe, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community

Figure 2.1. Map showing location of study sites. Each point represents a group of three study sites including a pollinator garden, an ornamental garden, and a lawn. Sites are located on an urban-rural gradient in the greater Chicago area IL, USA.

Table 2.1. List of all species collected in bee bowls. Sampling occurred in registered pollinator gardens, ornamental gardens, and lawns in the greater Chicago area IL, USA between June and August in 2018 and 2019. Superscripts indicate references (Appendix B).

Species	Number Collected	Percent Abundance	Family	Origin	Feeding
Andrena commoda	1	0.11	Andrenidae	Native[13]	Polylectic[3,13]
Andrena hippotes	2	0.22	Andrenidae	Native[13]	Polylectic[13]
Andrena wilkella	2	0.22	Andrenidae	Exotic[13]	Polylectic[11,13]
Calliopsis andreniformis	25	2.78	Andrenidae	Native[22]	Polylectic[8,22]
Anthophora terminalis	3	0.33	Apidae	Native[14]	Polylectic[13,14]
Apis mellifera	18	2.00	Apidae	Exotic[13]	Polylectic[8.13]
Bombus bimaculatus	2	0.22	Apidae	Native[13]	Polylectic[10,13]
Bombus citrinus	1	0.11	Apidae	Native[6]	Polylectic[7]
Bombus fervidus	1	0.11	Apidae	Native[13]	Polylectic[8,13]
Bombus impatiens	1	0.11	Apidae	Native[13]	Polylectic[8,13]
Ceratina calcarata	6	0.67	Apidae	Native[13]	Polylectic[8,13]
Ceratina mikmaqi	2	0.22	Apidae	Native[6]	Polylectic[8,10]
Ceratina strenua	8	0.89	Apidae	Native[13]	Polylectic[8,13]
Eucera hamata	1	0.11	Apidae	Native[13]	Polylectic[8,13]
Melissodes agilis	14	1.56	Apidae	Native[13]	Oligolectic[8,13]
Melissodes bimaculatus	188	20.94	Apidae	Native[13]	Polylectic[8,13]
Melissodes desponsa	1	0.11	Apidae	Native[13]	Oligolectic[8,13]
Melissodes subillatus	2	0.22	Apidae	Native[13]	Polylectic[8,13]
Melissodes trinodis	6	0.67	Apidae	Native[13]	Polylectic[8,13]
Melissodes vernoniae	1	0.11	Apidae	Native[7]	Oligolectic[7,12]
Peponapsis pruinosa	3	0.33	Apidae	Native[13]	Oligolectic[13]
Hylaeus affinis/modestus	4	0.45	Colletidae	Native[13]	Polylectic[13]
Hylaeus hyalinatus	1	0.11	Colletidae	Exotic[6,2]	Polylectic[2]
Hylaeus illinoisensis	3	0.33	Colletidae	Native[6]	Polylectic[10,17]
Hylaeus mesillae	6	0.67	Colletidae	Native[13]	Polylectic[13,18]
Agapostemon virescens	124	13.81	Halictidae	Native[13]	Polylectic[8,13]
Augochlora pura	18	2.00	Halictidae	Native[13]	Polylectic[8,13]
Augochlorella aurata	5	0.56	Halictidae	Native[13]	Polylectic[8,13]
Augochlorella persimilis	3	0.33	Halictidae	Native[7]	Polylectic[8,19]
Augochloropsis fulgida	1	0.11	Halictidae	Native[8,10]	Polylectic[6,7]
Halictus confusus	51	5.68	Halictidae	Native[13]	Polylectic[13]
Halictus ligatus	53	5.90	Halictidae	Native[13]	Polylectic[8,13]
Halictus parallelus	1	0.11	Halictidae	Native[6]	Polylectic[1,7]
Halictus rubicundus	5	0.56	Halictidae	Native[13]	Polylectic[8,13]
Lasioglossum "dark admirandum"	1	0.11	Halictidae	Native[13]	Polylectic[6,13]
Lasioglossum acuminatum	3	0.33	Halictidae	Native[13]	Polylectic[13]
Lasioglossum admirandum	18	2.00	Halictidae	Native[6,13]	Polylectic[10,13]
Lasioglossum albipenne	1	0.11	Halictidae	Native[6]	Polylectic[7,10]
Lasioglossum anomalum	23	2.56	Halictidae	Native[6,13]	Polylectic[10,13]
Lasioglossum bruneri	1	0.11	Halictidae	Native[6,13]	Polylectic[10,13]
Lasioglossum cinctipes	13	1.45	Halictidae	Native[13]	Polylectic[10,13]
Lasioglossum coeruleum	2	0.22	Halictidae	Native[23]	Polylectic[10,18]
Lasioglossum coriaceum	6	0.67	Halictidae	Native[13]	Polylectic[10,13]
Lasioglossum cressonii	10	1.11	Halictidae	Native[13]	Polylectic[13,18]
Lasioglossum forbesii	1	0.11	Halictidae	Native[6]	Polylectic[9]
Lasioglossum hitchensi	85	9.47	Halictidae	Native[6,13]	Polylectic[13,18]

Table 2.1. cont.

Lasioglossum illinoense	21	2.34	Halictidae	Native[6,13]	Polylectic[10,13]
Lasioglossum imitatum	6	0.67	Halictidae	Native[13]	Polylectic[10,13]
Lasioglossum leucozonium	1	0.11	Halictidae	Exotic[6,13]	Polylectic[13]
Lasioglossum macoupinense	3	0.33	Halictidae	Native[6]	Polylectic[4,10]
Lasioglossum michiganensis	1	0.11	Halictidae	Native[6,13]	Polylectic[13]
Lasioglossum obscurum	1	0.11	Halictidae	Native[6]	Polylectic[10,14]
Lasioglossum oenotherae	5	0.56	Halictidae	Native[6,13]	Oligolectic[13]
Lasioglossum paradmirandum	10	1.11	Halictidae	Native[6]	Polylectic[7,10]
Lasioglossum pectorale	10	1.11	Halictidae	Native[6,13]	Polylectic[10,13]
Lasioglossum pilosum	1	0.11	Halictidae	Native[6,13]	Polylectic[13,18]
Lasioglossum platyparium	1	0.11	Halictidae	Native[6,13]	Polylectic[13]
Lasioglossum pruinosum	3	0.33	Halictidae	Native[7]	Polylectic[7,10]
Lasioglossum gotham group	10	1.11	Halictidae	Native[13]	Polylectic[10,13]
Lasioglossum spp.	13	1.45	Halictidae		
Lasioglossum tegulare	10	1.11	Halictidae	Native[6,13]	Polylectic[10,13]
Lasioglossum truncatum	2	0.22	Halictidae	Native[6]	Polylectic[16]
Lasioglossum versans	1	0.11	Halictidae	Native[6,13]	Polylectic[13]
Lasioglossum versatum	1	0.11	Halictidae	Native[6,13]	Polylectic[13,18]
Lasioglossum zephyrum	48	5.35	Halictidae	Native[7]	Polylectic[10,18]
Lasioglossum zonulum	2	0.22	Halictidae	Exotic[6]	Polylectic[5]
Anthidium manicatum	3	0.33	Megachalidae	Exotic[13]	Polylectic[10,13]
Anthidium oblongatum	1	0.11	Megachalidae	Exotic[13]	Polylectic[13]
Chelostoma philadelphi	1	0.11	Megachalidae	Native[6]	Oligolectic[21]
Chelostoma rapunculi	1	0.11	Megachalidae	Exotic[6]	Oligolectic[15]
Heriades carinatus	1	0.11	Megachalidae	Native[6]	Polylectic[9]
Hoplitus producta	1	0.11	Megachalidae	Native[13]	Polylectic[13]
Megachile campanulae	3	0.33	Megachalidae	Native[6]	Polylectic[10,20]
Megachile petulans	2	0.22	Megachalidae	Native[6]	Polylectic[9,10]
Megachile rotundata	6	0.67	Megachalidae	Exotic[13]	Polylectic[10,13]
Unknown	2	0.22			

Table 2.2. Parameter estimates for all competitive models including flower density and urbanization as independent variables and pollinator abundance or richness as a dependent variable. Standard errors are in parentheses. The symbol ‡ denotes parameters that are considered to be informative and were used to calculated predicted values used to generate figures.

	Intercept	**Site Area**	**Year**	**Day**	**Minute**	**Temperature**	**Wind Speed**	**Cloud Cover**	**Flower Density**	**Urbanization**	**Flower Density*Urbanization**	**ΔAIC_C**	**AIC_C Weight**	**Log-Likelihood**
Honey Bee Abundance (Surveys)														
Model 1	-3.06 (0.83)			0.018‡ (0.0036)					0.0011‡ ($3.2*10^{-4}$)	0.024‡ (0.0093)		0.00	0.54	-616.12
Bumble Bee Abundance (Surveys)														
Model 1	-2.14 (-0.46)			0.015‡ (0.0023)					$3.4*10^{-4}$‡ ($2.1*10^{-4}$)			0.00	0.41	-466.58
Model 2	-2.15 (0.46)			0.015‡ (0.0023)								0.67	0.29	-467.95
Bumble Bee Richness (Surveys)														
Model 1	-0.87 (0.26)			0.0068‡ (0.0029)					$3.1*10^{-4}$‡ ($1.1*10^{-4}$)			0.00	0.43	-271.14
Total Bee Abundance (Surveys)														
Model 1	-4.56 (0.81			0.039‡ (0.0039)					$8.2*10^{-4}$‡ ($3.58*10^{-4}$)			0.00	0.05	-655.46
Total Bee Abundance (Bowls)														
Model 1	0.0013 (0.020)		-0.66‡ (0.10)						0.0015‡ ($4.28*10^{-4}$)	-0.0032 (0.0039)	$-2.3*10^{-5}$‡ ($8.3*10^{-6}$)	0.00	0.87	-500.06
Bee Richness (Bowls)														
Model 1	0.0012 (0.015)		-0.53‡ (0.073)						$7.2*10^{-4}$‡ ($3.1*10^{-4}$)	-0.0028 (0.0028)	$1.2*10^{-5}$‡ ($6.0*10^{-6}$)	0.00	0.43	-382.02
Model 2	1106.5 (147.7)		-0.55‡ (0.07)									0.23	0.66	-385.75

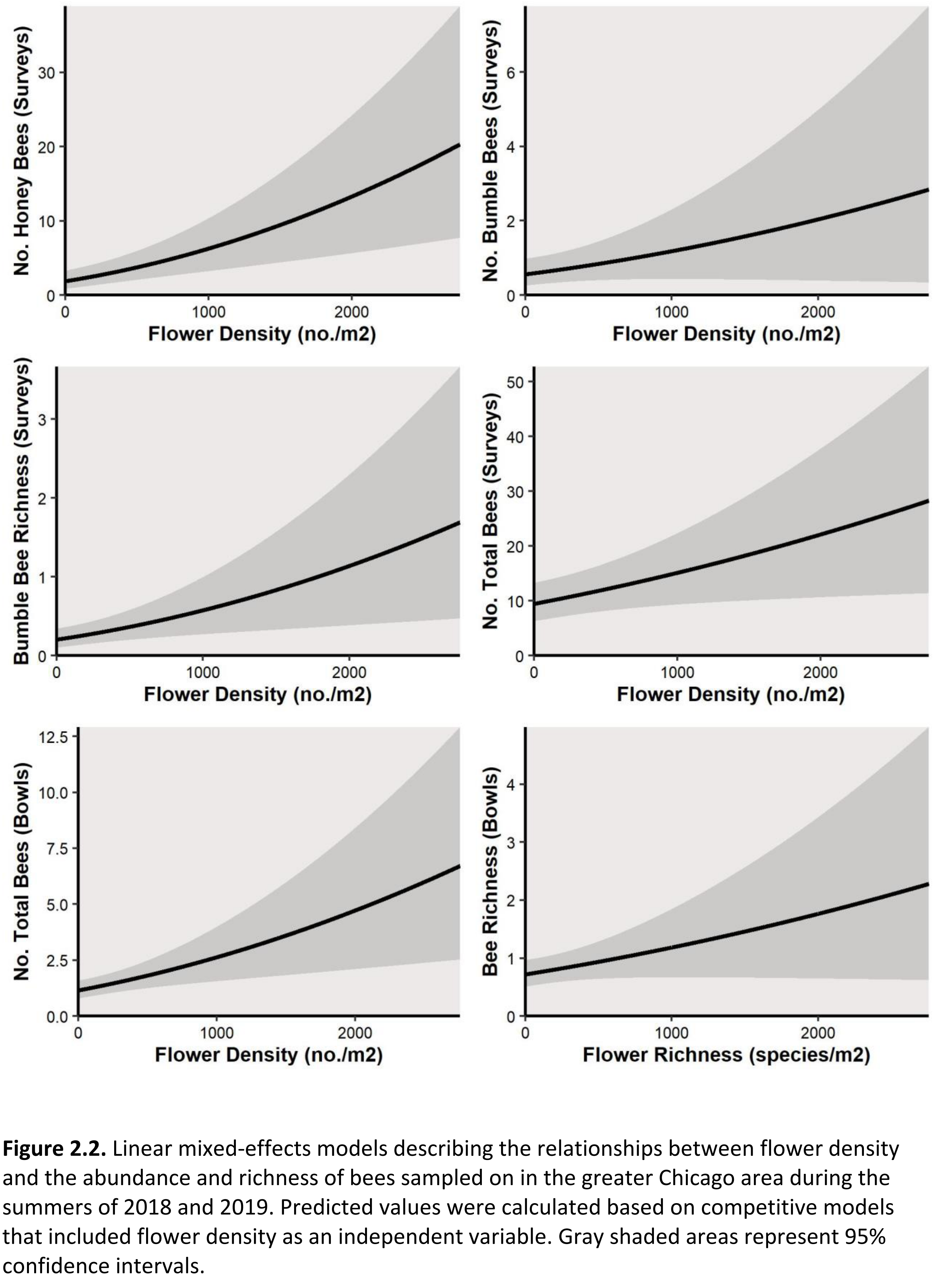


Figure 2.2. Linear mixed-effects models describing the relationships between flower density and the abundance and richness of bees sampled on in the greater Chicago area during the summers of 2018 and 2019. Predicted values were calculated based on competitive models that included flower density as an independent variable. Gray shaded areas represent 95% confidence intervals.

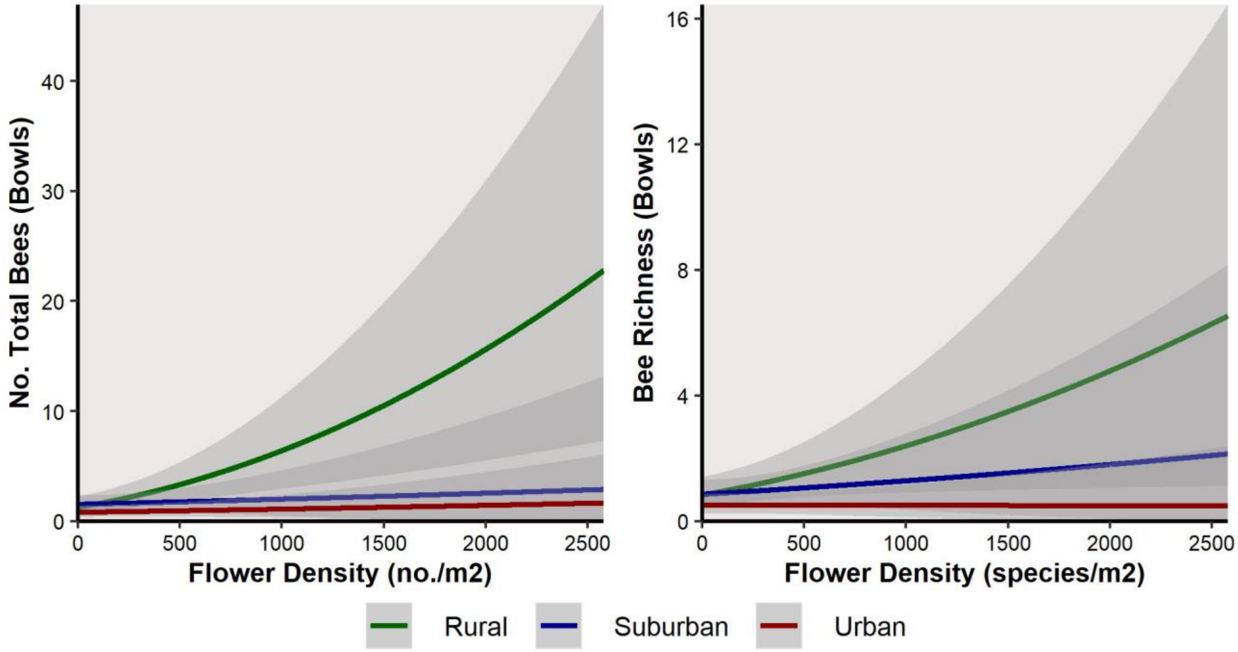


Figure 2.3. Linear mixed-effects models including the interaction effects between flower density and urbanization on the abundance and richness of bees collected in gardens and lawns in the greater Chicago area during the summers of 2018 and 2019. Predicted values were calculated based on competitive models that included the interaction between flower density and urbanization. Gray shaded areas represent 95% confidence intervals.

Table 2.3. Parameter estimates for all competitive models including flower richness and urbanization as independent variables and pollinator abundance or richness as a dependent variable. Standard errors are in parentheses. The symbol ‡ denotes parameters that are considered to be informative and were used to calculated predicted values used to generate figures.

	Intercept	Site Area	Year	Day	Minute	Temperature	Wind Speed	Cloud Cover	Flower Richness	Urbanization	Flower Richness*Urbanization	ΔAIC_C	AIC_C Weight	Log-Likelihood
Honey Bee Abundance (Surveys)														
Model 1	-2.58 (0.88)			0.016‡ (0.0037)					-0.29 (4.09)	0.014 (0.011)	0.13‡ (0.085)	0.00	0.49	-616.49
Model 2	-2.99 (0.84)			0.015‡ (0.0037)					5.35‡ (1.74)	0.024‡ (0.0098)		0.23	0.44	-617.65
Bumble Bee Abundance (Surveys)														
Model 1	-2.01 (0.43)			0.011‡ (0.0022)					6.46‡ (0.71)			0.00	0.42	-436.19
Model 2	-2.02 (0.47)			0.011‡ (0.0022)					4.13‡ (1.69)	$-5.9*10^{-4}$ $(4.4*10^{-3})$	0.052‡ (0.036)	0.74	0.29	-434.48
Bumble Bee Richness (Surveys)														
Model 1	-0.85 (0.27)			0.0053‡ (0.0013)					1.68‡ (1.00)	$2.4*10^{-4}$ $(2.6*10^{-3})$	0.032‡ (0.021)	0.00	0.37	-251.58
Model 2	-0.95 (0.27)			0.0053‡ (0.0013)					3.08‡ (0.41)	0.0029‡ (0.0019)		0.24	0.33	-252.76
Model 3	-0.81 (0.25)			0.0052‡ (0.0013)					3.10‡ (0.42)			0.34	0.31	-253.84
Total Bee Abundance (Surveys)														
Model 1	-4.35 (0.77)			0.033‡ (0.0039)					11.56‡ (1.87)			0.00	0.63	-640.42

Table 2.3 cont.

Total Bee Abundance (Bowls)								
Model 1	1371 (206)	-0.68‡ (0.10)	1.22‡ (0.82)			0.00	0.26	-506.00
Model 2	1326 (207)	-0.66‡ (0.10)	4.31‡ (2.23)	$-1.53*10^{-4}$ ($5.42*10^{-3}$)	-0.071‡ (0.048)	0.08	0.25	-503.96
Model 3	1398 (207)	-0.69‡ (0.10)				0.15	0.25	-507.10
Bee Richness (Bowls)								
Model 1	1106 (148)	-0.55‡ (0.073)				0.00	0.29	-385.75
Model 2	1083 (148)	-0.54‡ (0.073)	0.80 (0.56)	-0.0041‡ (0.0028)		0.17	0.27	-383.78

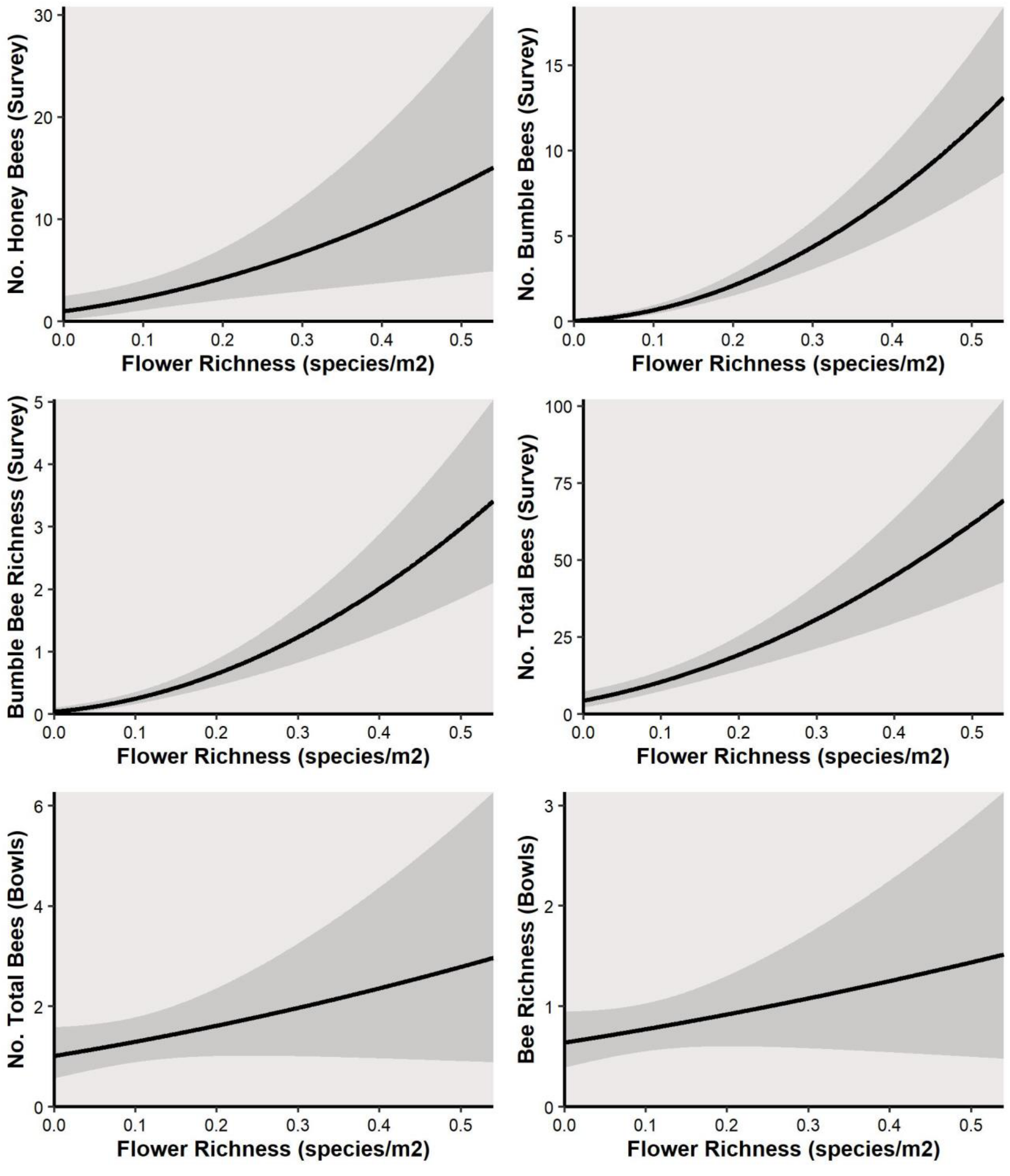


Figure 2.4. Linear mixed-effects models describing the relationships between flower richness and the abundance and richness of bees sampled in the greater Chicago area during the summers of 2018 and 2019. Predicted values were calculated based on competitive models that included flower richness as an independent variable. Gray shaded areas represent 95% confidence intervals.

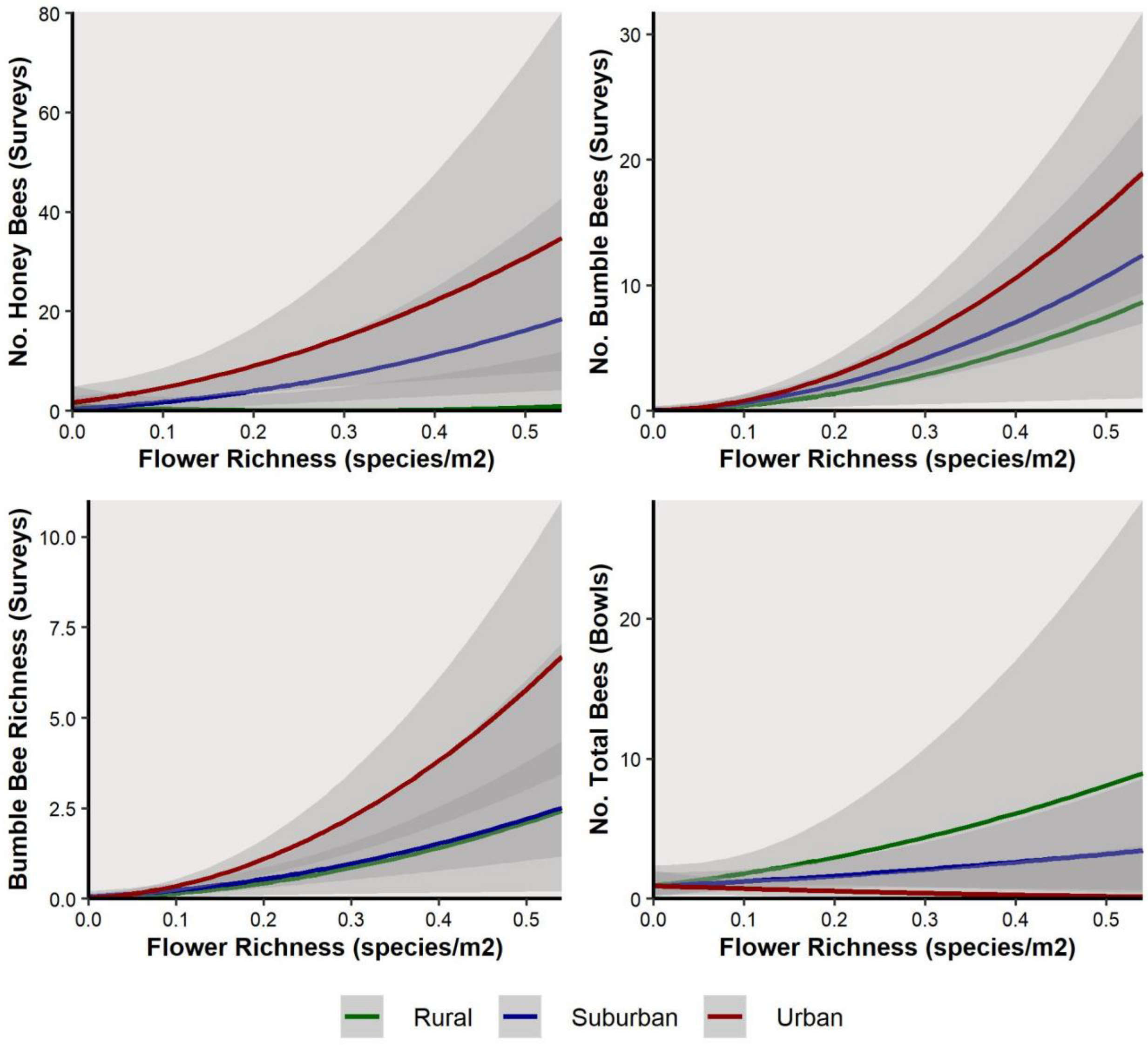


Figure 2.5. Linear mixed-effects models including the interaction effects between flower richness and urbanization on the abundance and richness of bees collected in gardens and lawns in the greater Chicago area during the summers of 2018 and 2019. Predicted values were calculated based on competitive models that included the interaction between flower richness and urbanization. Gray shaded areas represent 95% confidence intervals.

Table 2.4. Parameter estimates for all competitive models including site type and urbanization as independent variables and pollinator abundance or richness as a dependent variable. Standard errors are in parentheses. The symbol ‡ denotes parameters that are considered to be informative and were used to calculated predicted values used to generate figures.

	Intercept	Site Area	Year	Day	Minute	Temperature	Wind Speed	Cloud Cover	Site Type	Urbanization	Urbanization^2	Site Type*Urbanization	Site Type*Urbanization^2	ΔAIC_C	AIC_C Weight	Log-Likelihood
Honey Bee Abundance (Surveys)																
Model 1	-4.53 (0.97)			0.018‡ (0.0036)					0.71‡ (0.25)	0.026‡ (0.0089)				0.00	0.37	-618.77
Bumble Bee Abundance (Surveys)																
Model 1	-3.48 (0.49)			0.015‡ (0.0023)					0.65‡ (0.093)	2.43‡ (1.50)	-2.25‡ (1.45)			0.00	0.23	-450.77
Model 2	-3.67 (0.52)			0.015‡ (0.0023)					0.66‡ (0.096)	0.0050‡ (0.0034)				0.26	0.20	-451.94
Model 3	-3.42 (0.49)			0.015‡ (0.0023)					0.66‡ (0.099)					0.34	0.19	-453.02
Bumble Bee Richness (Surveys)																
Model 1	-1.47 (0.33)			0.0071‡ (0.0013)					0.22‡ (0.10)	-0.0040 (0.0044)		0.0038‡ (0.0021)		0.00	0.37	-254.99
Model 2	-1.79 (0.29)			0.0071‡ (0.0013)					0.038‡ (0.051)	0.0036‡ (0.0018)				1.17	0.21	-256.62
Total Bee Abundance (Surveys)																
Model 1	-7.58 (0.92)			0.040‡ (0.0039)					1.53‡ (0.24)					0.00	0.44	-644.48
Total Bee Abundance (Bowls)																
Model 1	1388 (206)		-0.69‡ (0.10)						0.25‡ (0.11)					0.00	0.31	-504.66

Table 2.4 cont.

Bee Richness (Bowls)								
Model 1	1101 (147)		-0.55‡ (0.073)		0.18‡ (0.076)	0.00	0.31	-383.20
Flower Density								
Model 1	-1.46 (2.02)	-0.0014‡ (00012)			4.68‡ (0.84)			
Flower Richness								
Model 1	-0.15 (-0.055)			$8.4*10^{-4}$‡ ($9.1*10^{-5}$)	0.12‡ (0.024)			

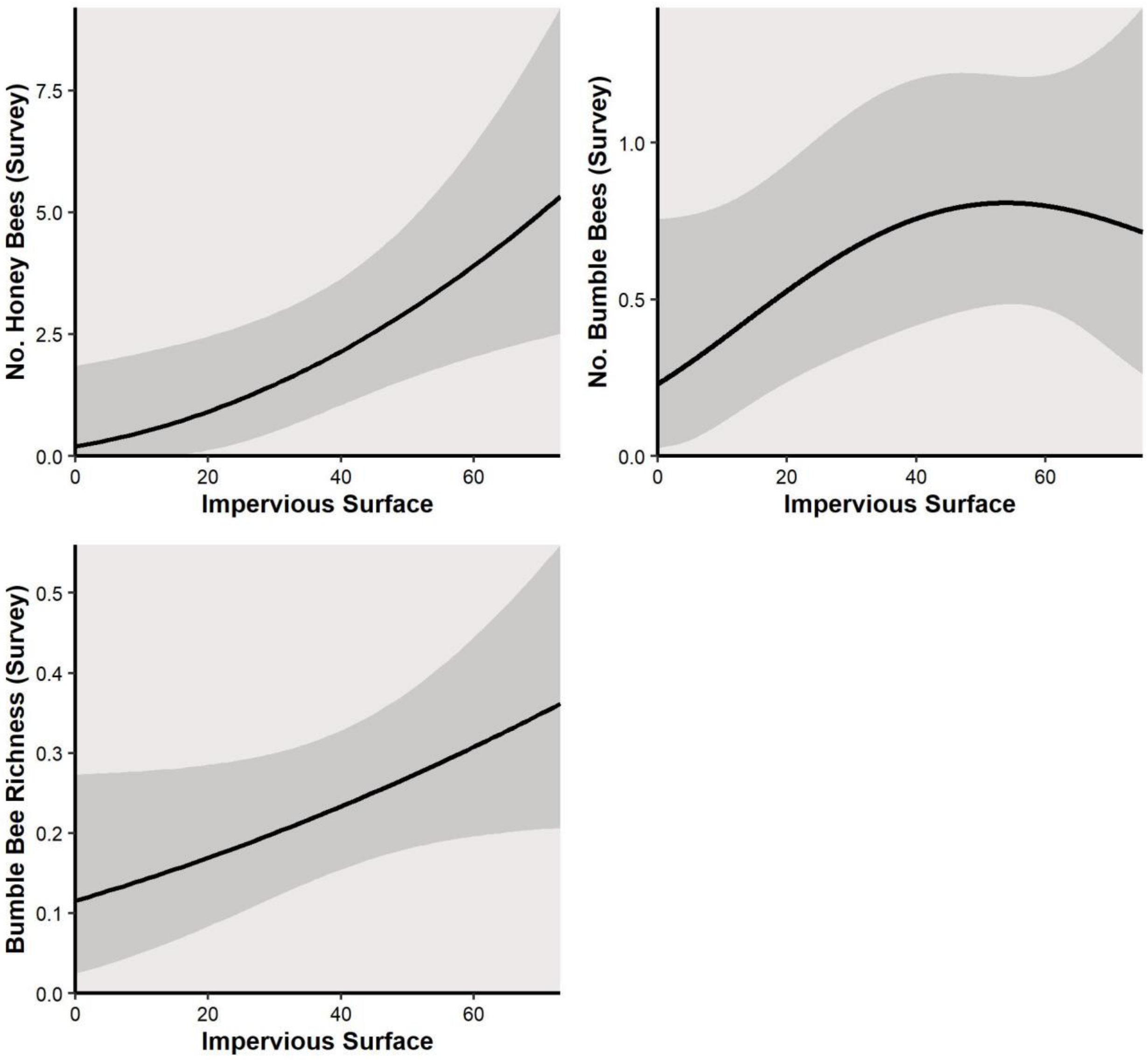


Figure 2.6. Linear mixed-effects models describing the relationships between impervious surface cover and the abundance and richness of bees sampled in the greater Chicago area during the summers of 2018 and 2019. Impervious surface cover was measured within a 2-km radius surrounding each study site. Predicted values were calculated based on competitive models that included impervious surface cover as an independent variable. Gray shaded areas represent 95% confidence intervals.

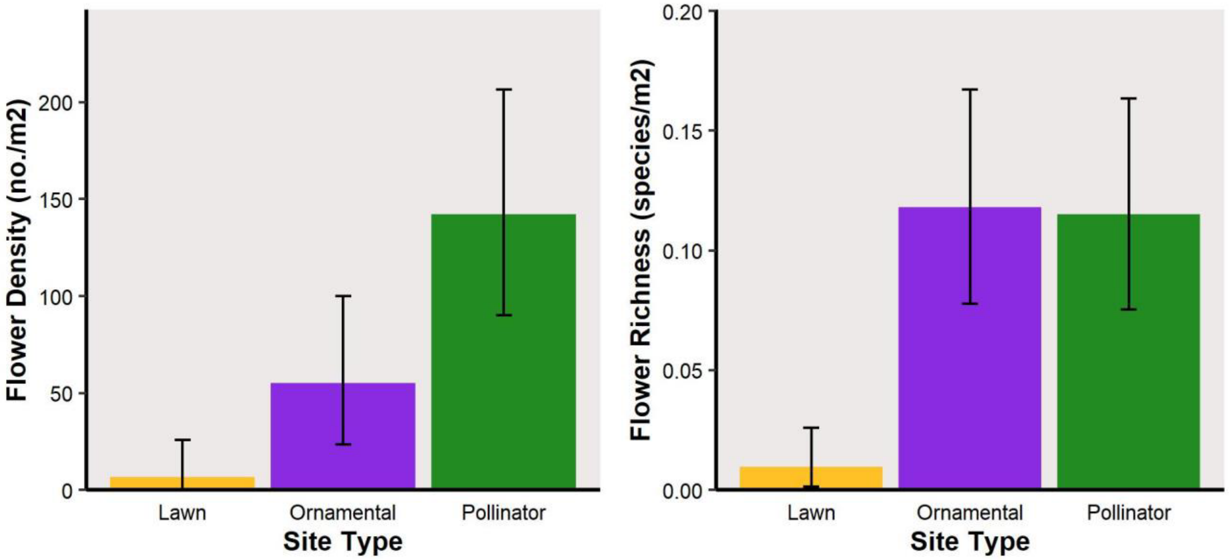


Figure 2.7. Bar plots showing the density and richness of flowers observed in pollinator gardens, ornamental gardens, and lawns. Data was collected in the greater Chicago area during the summers of 2018 and 2019. Predicted values were calculated based on competitive models that included site type as an independent variable. Error bars represent 95% confidence intervals.

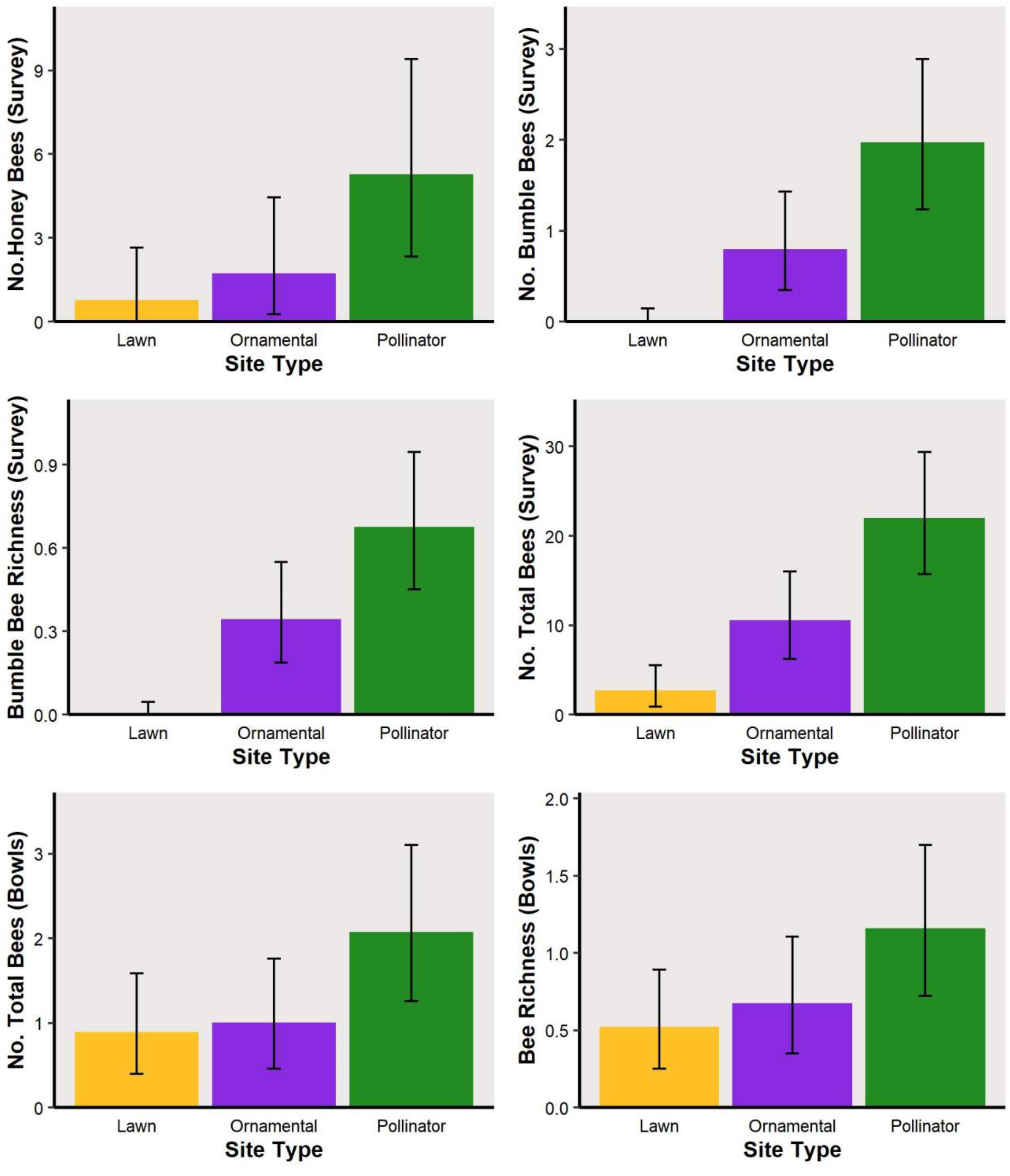


Figure 2.8. Bar plots showing the abundance and richness of bees observed in pollinator gardens, ornamental gardens, and lawns. Data was collected in the greater Chicago area during the summers of 2018 and 2019. Predicted values were calculated based on competitive models that included site type as an independent variable. Error bars represent 95% confidence intervals.

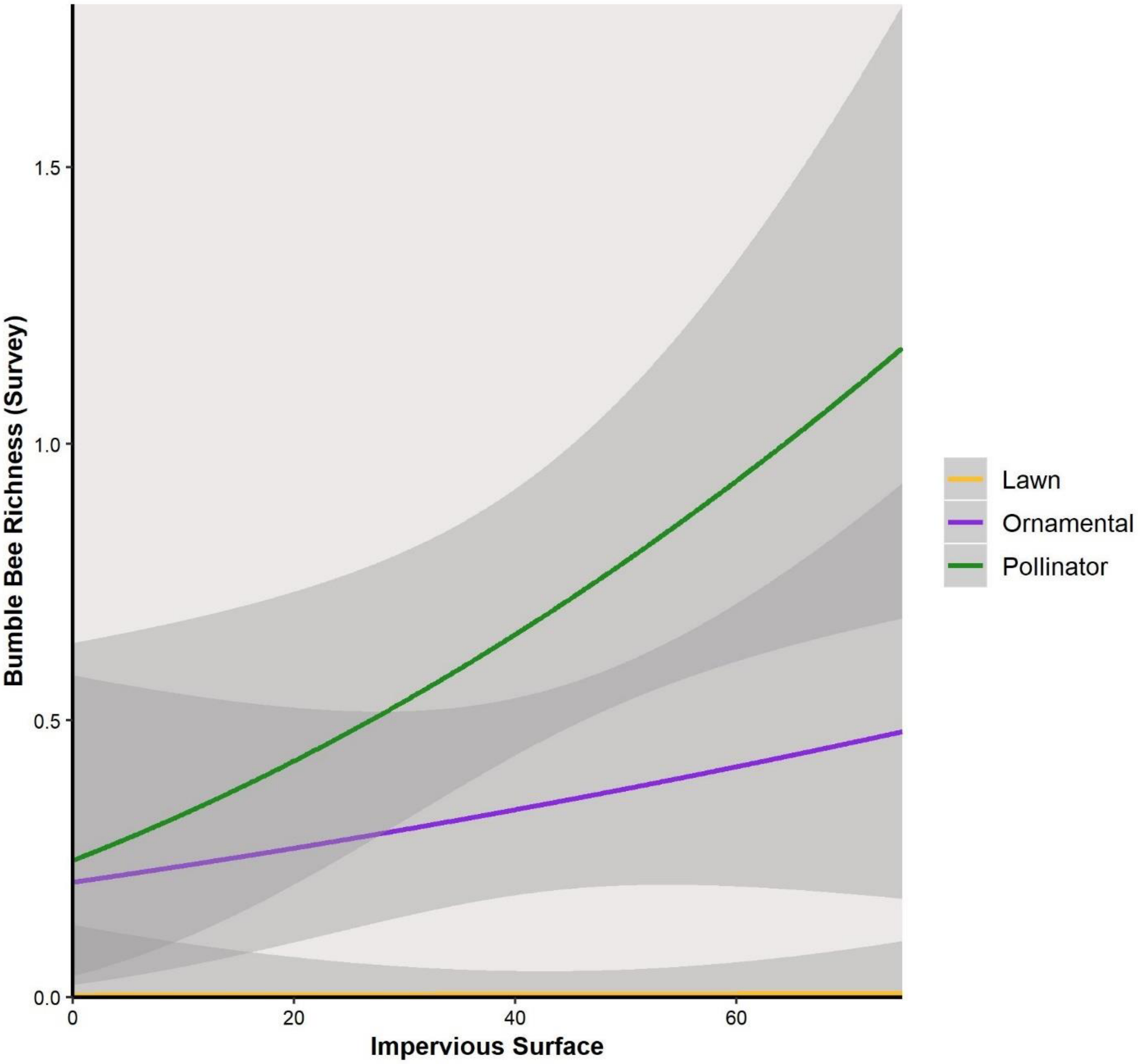


Figure 2.9. Linear mixed-effects models including the interaction effects between site type and urbanization on the richness of bumble bees collected in gardens and lawns in the greater Chicago area during the summers of 2018 and 2019. Predicted values were calculated based on competitive models that included the interaction between site type and urbanization. Gray shaded areas represent 95% confidence intervals.

CHAPTER 3: PEOPLE OF COLOR EXPERIENCE REDUCED ACCESS TO POLLINATORS ALONG AN URBAN-RURAL GRADIENT[2]

Abstract

Inequities based on racism and classism play an important role in shaping patterns of urban land cover and ecological functioning. Resulting disparities in access to important health, educational, and public safety benefits of green space threaten the well-being of city-dwellers, and potential negative impacts on urban pollinators are especially alarming given the essential ecosystem services that they provide. We investigated the relationships among income, race, historical discriminatory practices, and pollinator abundance along an urban-rural gradient in Illinois, USA. We did not observe a relationship between income and pollinator abundance. We did observe fewer pollinators in neighborhoods where more people of color live and for most pollinator groups this pattern was consistent along the urbanization gradient. We may be able to begin to correct these inequities by addressing societal structures that maintain systemic racism. Doing so will help us move towards a social-ecological system in which all people benefit from the ecosystem services provided by pollinators.

Introduction

Access to nature has wide-ranging and profound impacts on human well-being. Interacting with green space and biodiversity improves physical and mental health (Marselle et al., 2021; Sandifer et al., 2015), crime rates are lower in neighborhoods with more green space (Kuo &

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[2] L.R. Lynch and J.R. Miller. Prepared for submission to *Urban Ecosystems*

Sullivan, 2001; Shepley et al., 2019), and green space provides opportunities for hands-on science education (Saunders et al., 2018; Wolsink, 2015). Additionally, insect pollinators help to support food security in urban neighborhoods where residents sometimes make use of gardening as a supplemental source of food and income (Abdulkadir et al., 2012).

Considering these benefits, it is clear that the well-being of city residents is likely to be negatively impacted by inequities in access to nature. Unfortunately, access to nature is often uneven, depending on social factors including income and race. For example, a "luxury effect," in which higher-income neighborhoods have greater access to biodiversity, is frequently observed (Hope et al., 2003b; Leong et al., 2018; Schell et al., 2020). The luxury effect is thought to occur because wealthier individuals and municipalities have more financial resources to direct towards gardening, green space management, and biodiversity conservation. Because human activity tends to have a particularly strong influence on ecosystem structure and function in urban areas, it might be expected that the luxury effect would also be particularly important there, and most evidence of the luxury effect has been documented in cities (Leong et al., 2018).

In addition to the impacts of the luxury effect, there is also evidence of inequities in access to nature based on race (Flocks et al., 2011; Lerman & Warren, 2011; Schell et al., 2020). Neighborhoods where more people of color live often have a reduced abundance of green space, less tree cover, and fewer native bird species when compared to predominantly white neighborhoods (Flocks et al., 2011; Lerman & Warren, 2011; Saporito & Casey, 2015). These patterns are likely explained in part by the legacies of discriminatory practices, and there have been recent calls for urban ecologists to explore the ways that historical racist policies continue to impact urban biodiversity today (Schell et al., 2020). For example, there is evidence that the practice of redlining, which occurred between 1935 and 1968, continues to impact the structure of urban ecosystems (Locke et al., 2021; Namin et al., 2020; Nardone et al., 2021; Schell et al., 2020).

Redlining was a form of legal discrimination in which the Home Owner's Loan Corporation (HOLC) used information including race and ethnicity to determine who should receive loans to buy or refinance their homes. Neighborhoods where people of color lived were labeled "hazardous," and residents of those neighborhoods were generally denied loans. As a result, these people were less likely to be able to own their own homes, many landlords abandoned their properties due to a lack of financial opportunities, and property values dropped (Rothstein, 2017). Because property taxes provide a large portion of funding for the development and maintenance of green spaces, this practice likely led to a reduction of habitat quantity as well as quality for many species in historically redlined areas (Schell et al., 2020).

In this study, we consider the relationships among income, race, historic redlining practices and the abundance of pollinators along an urban-rural gradient. While there is some evidence that the luxury effect impacts arthropods, including pollinators, little research has considered potential relationships with race and discriminatory practices (Baldock et al., 2019; Leong et al., 2016). We predicted a positive relationship between income and pollinator abundance, and a negative relationship between the proportion of people of color living in a neighborhood and pollinator abundance. We expected these patterns to be stronger in urban areas than in suburban or rural areas. Additionally, we predicted that we would observe lower pollinator abundance in historically redlined areas. Understanding these relationships will help us develop strategies that support pollinator conservation and environmental justice by correcting social inequities that may be negatively impacting pollinators. Correcting inequities in access to pollinator diversity is of particular importance given the essential ecosystem services that they provide as well as their charismatic nature which makes their presence especially valuable to people engaging with local ecosystems.

Methods

Study Area Description

This study was conducted in Illinois, USA. Our study area encompasses Chicago; several smaller cities and towns including Springfield, Champaign-Urbana, and Peoria; and rural areas consisting largely of cropland (Figure 3.1). While historical ecosystems in Illinois included tallgrass prairie and forest, approximately 70% of land in the state is now being used for agricultural purposes, mainly intensively managed corn and soy (Dewitz, 2019; US Department of Agriculture, 2015). Our study sites were not located on a linear gradient extending to a single large city, but rather distributed across a landscape (McDonnell & Hahs, 2008). We used impervious surface cover as a metric of urbanization.

Pollinator Sampling Methods

Data for this study were collected by community scientists between 30 May and 28 October 2020. Community science, also called “citizen science,” refers to the participation of people who are not professional scientists in academic research (Kovaka, 2021; Tulloch et al., 2013). Recently, a number of organizations supporting public engagement in scientific research have transitioned to the term “community science” to acknowledge that participants are not required to be citizens of any particular country (Fuller, 2020; National Audubon Society, 2022). Previous research suggests that community scientists can categorize pollinators into broad taxonomic groups at high levels of accuracy and detect important trends in pollinator visitation, especially when provided with direct training (Kremen et al., 2011; Ratnieks et al., 2016; Roy et al., 2016). We recruited individuals by contacting gardening organizations, extension offices, teachers at K-12 schools, university students, and people who had previously been involved in our research. Participation in the project was open to all interested people living in the state of Illinois. Some volunteers began with extensive knowledge of pollinator ecology and identification, whereas the topic was completely new to others.

Before beginning data collection, all community scientists participated in a two-hour online training. Thirteen training sessions facilitated by L. Lynch were held in late May and early June 2020, with a total of 161 participants. The training included an introduction to the topic of pollinator conservation, an explanation of the data collection procedure, information on data submission using our online portal, and an overview of key characteristics used to identify pollinators. Additionally, participants practiced identifying pollinators from photos in breakout rooms, attempted the data collection procedure in their own yards, and returned to the group online for a question-and-answer session. During the training, participants learned to classify pollinators into 13 broad groups: Honey Bees (species *Apis mellifera*), Bumble Bees (genus *Bombus*), Green Sweat Bees (a subset of genera within the Halictid family), Hairy Belly Bees (female Megachalids), Other Bees, Wasps, Flies, Butterflies, Moths, True Bugs, Beetles, Spiders, and Birds (modified from groups recommended for pollinator-focused community science projects by Jordan et al. 2016 and Kremen et al. 2011). We also offered an optional question-and-answer session in late July. This was an opportunity for participants to ask questions about pollinator identification or the data collection procedure that had arisen since the initial training.

After the training, community scientists selected at least one site in which to monitor pollinators throughout the summer. The site could be an ornamental garden, vegetable garden, or garden designed to provide wildlife habitat. Due to social distancing recommendations associated with the Covid-19 pandemic, we encouraged participants to select study sites located at or near their homes. Sites ranged in size from 0.9 to 46.5 m^2. Because insect pollinators make use of small patches of resources in residential areas and floral resource availability appears to have a stronger impact on pollinator visitation rates than patch size, we did not expect the small size of some sites to be a problem (Battle et al., 2021; Blaauw & Isaacs, 2014; Sowig, 1989). Pollinators were observed even at our smallest study sites, and although we tested site area as a potential covariate for each of our models, it was never found to be important.

Each participant monitored their site(s) throughout the summer and fall. While we instructed them to monitor sites every two weeks from June through October, we accepted data even when it was not collected regularly for the duration of the study. All pollinator observations were conducted between 0730 and 1800 h on non-rainy days with wind speeds < 6.2 km/h.

While hand-netting is the most common method used to sample pollinators actively, visual observation can be an effective alternative when it is necessary to avoid damage to ornamental plants (Matteson et al., 2013) or to facilitate the participation of community scientists (Kremen et al., 2011; Ratnieks et al., 2016; Roy et al., 2016). Participants spent 30 min walking slowly through their garden and recording all pollinators observed landing on reproductive parts of flowers. They visited all plants in the garden but spent the most time observing the most active flowers (modified from Hülsmann et al., 2015; Jordan et al., 2016). Pollinators were identified to the 13 broad groups listed above. If the same individual pollinator was observed visiting multiple flowers, it was only recorded once. However, participants did record individuals if they were unsure of whether they had already been observed (Hülsmann et al., 2015). At each visit, participants also recorded garden type (vegetable, ornamental, wildlife habitat), garden size, shade temperature, wind speed, and cloud cover.

Social and Environmental Variables

We used data from the 2019 American Community Survey (ACS) to calculate the proportion of the population within each census tract made up of people of color (Black or African American, American Indian or Alaska Native, Asian, Native Hawaiian or Pacific Islander, and multiracial; US Census Bureau, 2019). Median per capita income was also derived from the 2019 ACS. We used the Intersect Tool in ArcMap to extract information about race and income for each of our study sites (ESRI, 2011).

We used data from the NLCD Urban Imperviousness Map (Dewitz, 2019) to calculate the percent cover of impervious surface in the landscape surrounding each of our study sites. We measured impervious surface cover within the typical daily foraging distance for each of the pollinator groups. Because we combined or dropped several of the pollinator groups identified by community scientists to increase accuracy, or due to a lack of data, the groups considered in our analyses were bees, wasps, flies, butterflies and moths (lepidoptera), beetles, true bugs, and total pollinators. Foraging distances can vary widely between individual species. We therefore attempted to select scales that would represent an average individual belonging to each of the seven pollinator groups. For example, because the various species of bees that we expected to be most frequently observed typically travel anywhere from 100 m to 1.5 km each day (Carr-Markell et al., 2020; Greenleaf et al., 2007; Hamblin et al., 2017; Zurbuchen et al., 2010), we measured impervious surface cover within a 750-m radius circle surrounding each of our study sites. A scale of 250 m was used for flies (Bates et al., 2011) and true bugs (Philpott et al., 2013), 500 m for butterflies and moths (Perović et al., 2015; Schmidt & Roland, 2006; Uhl et al., 2020; Villemey et al., 2015), 750 m for wasps (Greenleaf et al., 2007), and 1 km for beetles (Delgado de la Flor et al., 2017). For models describing overall pollinator visitation, we used a scale of 500 m. We used the Buffer and Tabulate Area tools in ArcGIS (ESRI, 2011) to calculate the percent cover of impervious surface within each buffer.

To examine the relationship between pollinator visitation and the lagging impacts of historical housing discrimination practices, we used maps from the federal government's Home Owners' Loan Corporation (HOLC). Neighborhoods with grades of A were considered "minimal risk" by the HOLC and were able to easily access financing, whereas neighborhoods with grades of D were considered "hazardous" and were generally unable to obtain loans (Nelson et al., 2021).We used the Intersect Tool to extract information about the HOLC grade where each of our study sites was located (ESRI, 2011).

Analyses

We used an information-theoretic approach to identify factors impacting pollinator visitation rates. We first used AIC_C to identify covariates. For each pollinator group, we compared models including the individual effects of date, shade temperature, wind speed, could cover, and garden area on pollinator visitation. The covariates from all models ranking above the null whose cumulative weight added up to 0.95 were included (Burnham & Anderson, 2002). If a model's AIC_C value was between zero and two points lower than the value of the null model, we included the covariate only if the 85% confidence interval of it's parameter estimate did not overlap zero (Arnold, 2010). Each model also included study site as a random effect to correct for non-independence from repeated sampling (Harrison et al., 2018b). We used either a square root transformation or a log transformation for each model to meet assumptions of normality and constant variance.

AIC_C was then used to compare linear mixed-effects models including the main and interaction effects of income, race, and urbanization on pollinator visitation (Anderson et al., 2000). For each group of pollinators, we compared thirteen models. The first three models included the individual effects of income, race, or urbanization. The next four models included the additive effects of these three variables in all possible combinations. We also considered models including the following: interaction effects between income and urbanization, interaction effects between income and urbanization along with additive effects of race, interaction effects between race and urbanization, interaction effects between race and urbanization as well as additive effects of income, and interaction effects between income and urbanization along with interaction effects between race and urbanization. The last model considered was the null.

We considered all models within Δ 2 AIC_C of the top model to be competitive (Burnham and Anderson 2002; Table 1). We checked for potentially uninformative parameters by identifying models for which a simpler nested model was also included within the set of competitive models and that had an AIC_C value within two points of the nested model per additional parameter. Additionally, models that had an AIC_C value less than two points lower than the null model per additional parameter were considered to be potentially uninformative (Arnold, 2010). A model with potentially uninformative parameters was retained in the competitive model set if at least one of those parameters had an 85% confidence interval that did not overlap zero or if another model including only the potentially uninformative parameters was ranked above the null. Otherwise, the model was removed from the set of competitive models (Arnold, 2010).

We graphed predicted values for each pollinator group based on the parameters present within the top models whose 85% confidence interval did not overlap zero. When multiple competitive models included the same parameter, and the 85% confidence interval for the parameter estimate did not overlap zero in multiple models, we used model-averaging to generate predicted values for that parameter (Arnold, 2010; Burnham & Anderson, 2002).

We then created linear mixed-effects models describing the relationships between the HOLC zone in which a study site was located and present-day pollinator visitation. We used a Tukey's pairwise comparison test to determine whether present-day pollinator visitation differed significantly between gardens located in different HOLC grades. This analysis was conducted for the subset of study sites located in areas that had been redlined. All analyses were conducted in R (R Core Team, 2021).

Results

Community scientists collected a total of 270 pollinator surveys from 60 study sites. Impervious surface cover surrounding the sites ranged from 0.24% to 72.9% within a 1-km radius. Forty-two surveys from sixteen study sites were located in an area that had been categorized by the HOLC for the purpose of housing discrimination. Only one site was in an A-graded area, so we removed it from our analysis. Therefore, our results included only areas that had been assigned a grade of B, C, or D.

For all seven of the pollinator groups considered, the proportion of the population within a census tract made up of people of color was included as a variable in the competitive model set (Table 3.1). The average number of bees, butterflies and moths, flies, true bugs, beetles, and overall pollinators observed was lower in neighborhoods where more people of color lived (Figure 3.2). For these six groups, there was no evidence of interaction effects between race and urbanization. For wasps, the only competitive model did include interaction effects between race and urbanization. At urban and suburban study sites wasp visitation was lower in census tracts where more people of color lived, while at rural study sites visitation was higher in these census tracts (Figure 3.3)

For butterflies and moths, flies, and total pollinators, competitive models also included mean per capita income in the census tract where the study sites were located. Contrary to expectations, the number of butterflies and moths and the number of total pollinators was lower in wealthier areas. For these two groups, no interaction effects were observed between income and urbanization (Figure 3.4). For flies, the competitive set contained a model including the individual effects of income as well as a model including the interaction effects of income and urbanization. The model including the individual effects of income showed that fly abundance was positively correlated with income while the model including the interaction effects between income and urbanization showed that fly abundance increased with income only in urban gardens (Figures 3.4 and 3.5).

The relationship between impervious surface cover in the surrounding landscape and the number of pollinators observed was included in the set of competitive models for wasps, flies, true bugs, and beetles. The abundance of wasps, flies, and true bugs increased with urbanization while the abundance of beetles decreased with urbanization (Figure 3.6). For bees, butterflies and moths, and flies, the competitive model set also included the null (Table 3.1). See Appendix C for the complete AIC_C tables associated with these analyses.

For two out of four taxa considered, pollinator visitation was highest at study sites located in zones categorized as B by the HOLC and lowest in zones categorized as D. For bees, an average of 10.8 individuals were observed in B-graded areas, 6.1 in C-graded areas and 1.0 in D-graded areas. For total pollinators, 21.8 individuals were observed in B-graded areas, 19.8 in C-graded areas and 5.5 in D-graded areas. However, these differences were not found to be statistically significant. For butterflies and moths, an average of 0.4 individuals were observed in B-graded areas, 0.5 in C-graded areas, and 0.3 in D-graded areas, and for flies, an average of 2.3 individuals were observed in B-graded areas, 6.0 in C-graded areas, and 1.3 in D-graded areas (Figure 3.6).

Discussion

Our results provide some evidence that the luxury effect may be impacting the abundance of pollinating flies within our study area, and that this pattern may be strongest in urban areas. The relationship between income and abundance may be strongest in urban areas because plant communities in cities are often planted and carefully maintained by people. Wealthier families and neighborhoods may be able to devote more financial resources towards green space management, which may lead to a greater abundance and diversity of floral resources (Leong et al., 2018).

In contrast, we found that the abundance of butterflies and moths as well as overall pollinator abundance was lower in wealthier areas, with this pattern being consistent across the gradient of urbanization represented by our study sites. These results contrast with those of two recent studies that find bee species richness and overall pollinator abundance to be positively correlated with income (Baldock et al., 2019; MacIvor, 2015), as well as a third that finds no relationship between income and butterfly species richness (Keren et al., 2022). One potential explanation for these patterns could be increased intensity of green space management in wealthier neighborhoods. Due to an "ecology of prestige," (Grove et al., 2006) higher-income households often experience social pressure to maintain "neat" yards and have been found to spend more money maintaining their lawns and use more lawn chemicals (Fraser et al., 2013; Robbins et al., 2001; Zhou et al., 2009). More frequent mowing reduces the availability of floral resources while the use of pesticides and herbicides use may also directly impact pollinator health (Abraham et al., 2018; Lerman et al., 2018; Mengoni Goñalons & Farina, 2018). Additionally, wealthier neighborhoods may include fewer unmaintained green spaces such as vacant lots, which can provide habitat for pollinators (Sivakoff et al., 2018; Turo et al., 2020).

We also found that there were fewer pollinators in neighborhoods where more people of color live and that for most pollinator groups this pattern was consistent along an urban-rural gradient. Given that positive relationships between income and pollinator abundance were observed only for flies, this pattern does not appear to be strongly related to racial disparities in income. This result provides evidence that inequities in access to nature occur in rural as well as urban areas. Conversely, we found that the relationship between wasp abundance and the proportion of people of color living within a census tract was negative in urban and suburban areas but positive in rural areas.

More green space is often available in predominantly white neighborhoods (Jesdale et al., 2013; Saporito & Casey, 2015; Venter et al., 2020). However, the fact that we observed negative relationships between impervious surface cover and pollinator abundance only for beetles suggests that a reduced quantity of green space is likely not the main factor leading to racial inequities in pollinator abundance. Instead, this pattern may be due to a reduced quality of resources. Land use and land cover may not be a reliable indicator of resources and environmental conditions at a particular location (Miller & Hobbs, 2007), and the availability and quality of resources present within green spaces may be impacted by systemic racism. For example, bees rely on the availability of floral and nesting resources (Potts et al., 2003; Potts et al., 2005; Roulston & Goodell, 2011; Williams et al., 2015), the abundance and diversity of which could be reduced in parks receiving fewer financial resources. Alternatively, the higher concentrations of pollutants that are often present in neighborhoods inhabited by people of color (Bravo et al., 2016; Gray et al., 2013; Ringquist, 2005; Whitehead & Buchanan, 2019) could negatively impact pollinators (Sivakoff et al., 2020).

While the mechanisms leading to racial inequities in pollinator abundance are not yet clear, racist practices and policies may contribute to this pattern. Segregation has often caused people of color living in cities to be concentrated into areas where both the quantity of green space and the amenities available within public green space are reduced (Jesdale et al., 2013; Suminski et al., 2012). Evidence that cities spend less money on green spaces in neighborhoods where more people of color live and that relationships between race and the quantity of green space exist on public but not private land suggest that this pattern is due not to resident preferences, but rather to city policy and the decisions of public agents (Watkins & Gerrish, 2018; Wolch et al., 2005).

Housing discrimination has been identified as one factor that may contribute to inequities in access to biodiversity (Schell et al., 2020). Discriminatory housing practices prevent communities of color from accumulating wealth, even when residents' income matches that of white communities, by preventing the acquisition of property and other assets that pass from generation to generation (Maroto, 2016; Oliver & Shapiro, 2006). This reduced individual wealth, along with declines in property tax income that occur when landlords abandon properties in redlined areas, means that fewer resources are directed towards the development of green spaces (Schell et al., 2020). While our results appear to show patterns of reduced abundance of bees and total pollinators in redlined areas, these results are not significant. This may suggest that lagging impacts of historical discriminatory policies do not have a strong impact on pollinators. Pollinators are highly mobile and have short generation times, which may allow them to recover quickly from historical disturbances (Breland et al., 2018; Odanaka et al., 2019). It could be that present-day policies and practices have a stronger impact on pollinators. Finally, the small sample size for our analyses focused on redlining could have made it difficult to detect relationships with pollinator abundance.

Inequities in access to biodiversity are problematic because they prevent people of color from receiving the many benefits that can be gained from interacting with biodiversity and green space. Disparities in access to nature such as those that we have identified in this study may also contribute to disparities in health, public safety, and education. Pollinators are of particular importance because they are visible and charismatic and because they also provide essential ecosystem services. We may be able to reduce some of these inequities in access to biodiversity by addressing societal structures that maintain systemic racism. Doing so will help us move towards a future social-ecological system in which all people are able to benefit from the ecosystem services provided by pollinators.

Table and Figures

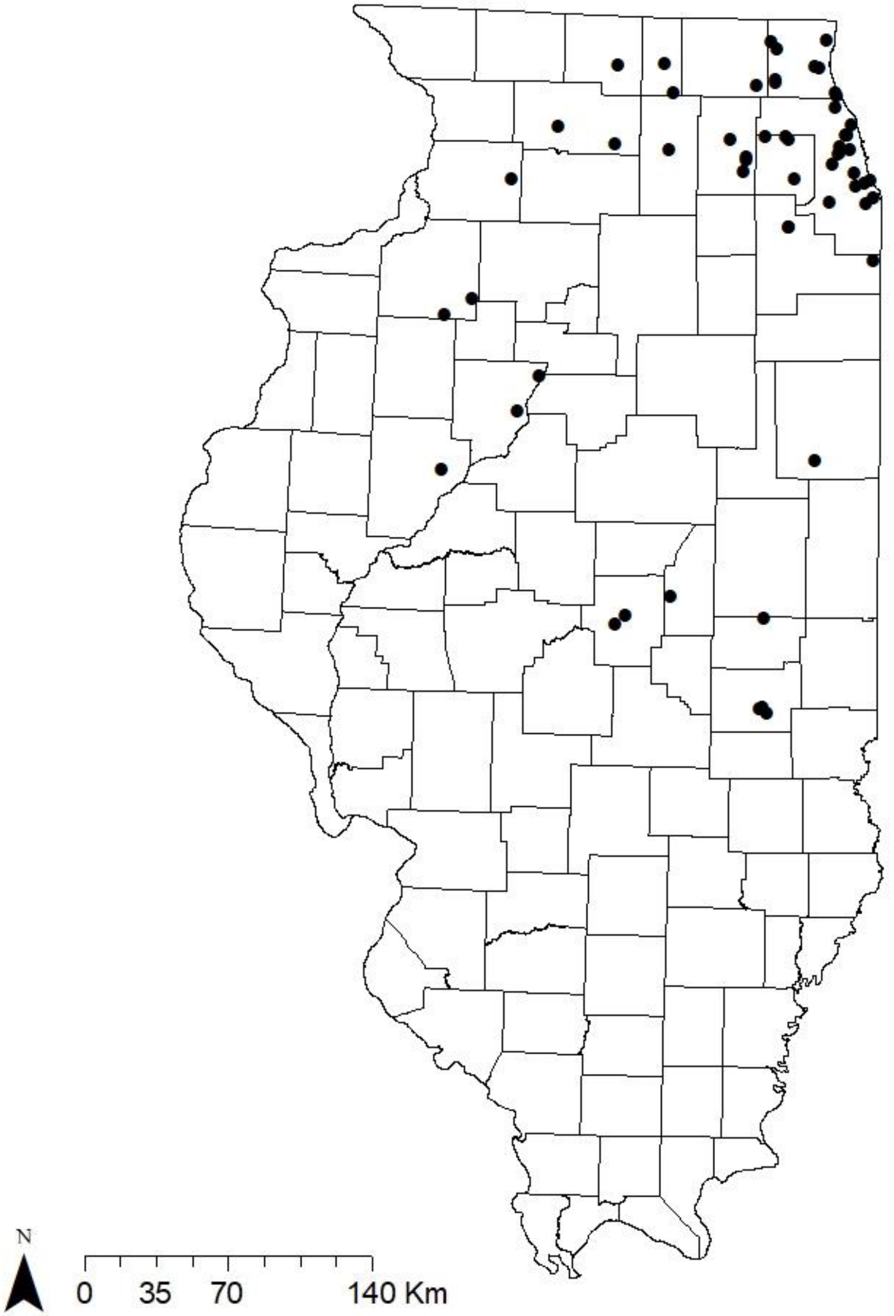

Figure 3.1. Map showing locations of study sites in Illinois, USA.

Table 3.1. Parameter estimates for all competitive models including race, income, and urbanization as independent variables and pollinator abundance as a dependent variable. Standard errors are in paranetheses. The symbol ‡ denotes parameters that are considered to be informative and were used to calculate predicted values used in figures.

	Intercept	Site Area	Day	Temperature	Wind Speed	Cloud Cover	Race	Income	Urbanization	Race*Urbanization	Income*Urbanization	ΔAIC_C	AIC_C Weight	Log-Likelihood
Beetle Abundance														
Model 1	-3.68 (0.93)			0.041‡ (0.011)			-0.024‡ (0.0075)					0.00	0.24	-528.58
Model 2	-3.43 (0.96)			0.041‡ (0.011)			-0.017‡ (0.0093)		-0.011‡ (0.010)			0.85	0.15	-527.96
Fly Abundance														
Model 1	-0.60 (1.58)		-0.0032 (0.0034)	0.020‡ (0.013)			-0.019‡ (0.0092)		0.019‡ (0.0095)			0.00	0.15	-534.86
Model 2	-0.68 (1.81)		-0.0026 (0.0034)	0.023‡ (0.013)				$1.3*10^{-5}$ ($2.1*10^{-5}$)	-0.025 (0.022)		$1.0*10^{-6\,\ddagger}$ ($5.9*10^{-7}$)	0.47	0.12	-534.03
Model 3	-0.31 (1.51)		-0.0035 (0.0034)	0.020 (0.013)								1.06	0.09	-537.49
Model 4	-1.07 (1.60)		-0.0031 (0.0034)	0.021‡ (0.013)				$1.64*10^{-5\ddagger}$ ($1.13*10^{-5}$)				1.07	0.09	-536.45
True Bug Abundance														
Model 1	0.030 (0.11)						-0.0089‡ (0.0034)		0.0052‡ (0.0034)			0.00	0.23	-309.50
Model 2	0.41 (0.088)						-0.0063‡ (0.0031)					0.09	0.22	-310.59
Wasp Abundance														
Model 1	-3.00 (0.77)		0.0070‡ (0.0029)				0.034 (0.025)		0.041‡ (0.013)	-0.011‡ (0.0047)		0.00	0.40	-519.87

Table 3.1 cont.

Lepidoptera Abundance								
Model 1	-4.92 (1.14)	0.014‡ (0.0023)	0.049‡ (0.0089)	-0.012‡ (0.0059)	$-1.4*10^{-5}$‡ ($8.1*10^{-6}$)	0.00	0.21	-429.59
Model 2	-6.07 (1.02)	0.015‡ (0.0023)	0.051‡ (0.0089)			0.73	0.15	-432.06
Bee Abundance								
Model 1	0.23 (0.92)		0.035‡ (0.12)			0.00	0.23	-537.85
Model 2	1.26 (1.13)		0.033‡ (0.012)	-0.014‡ (0.0096)	$-1.5*10^{-5}$ ($1.32*10^{-5}$)	1.54	0.11	-536.53
Pollinator Abundance								
Model 1	2.22 (1.36)		0.053‡ (0.014)	-0.028‡ (0.011)	$-2.4*10^{-5}$‡ ($1.5*10^{-5}$)	0.00	0.22	-588.32
Model 2	1.09 (1.16)		0.054‡ (0.014)	-0.021‡ (0.010)		0.37	0.18	-589.55

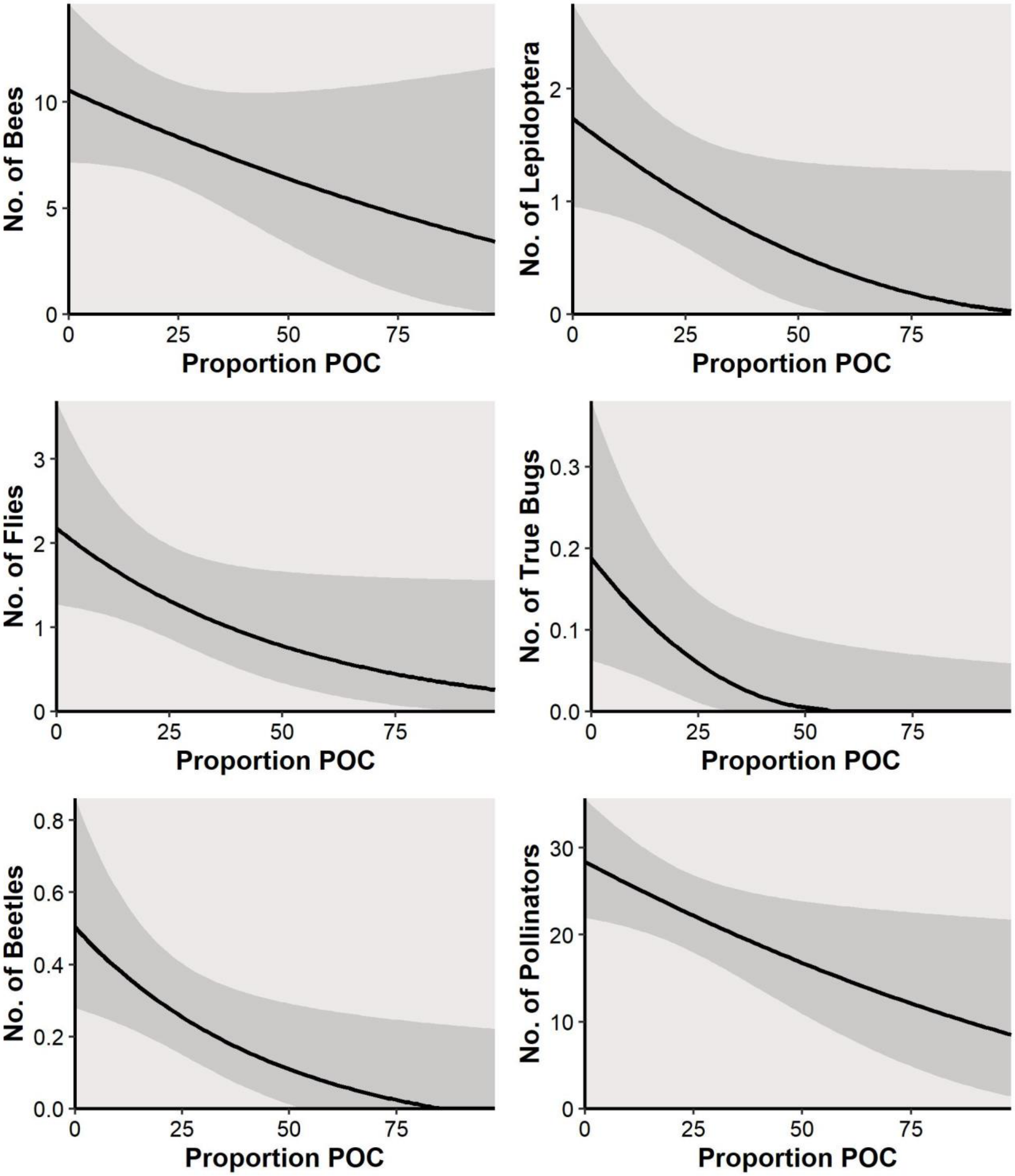

Figure 3.2. Relationships between the proportion of the population within a census tract made up of people of color (POC) and the average number of pollinators observed in Illinois gardens during the summer of 2020. Predicted values were calculated based on competitive models that included race as an independent variable. Gray shaded areas represent 95% confidence intervals.

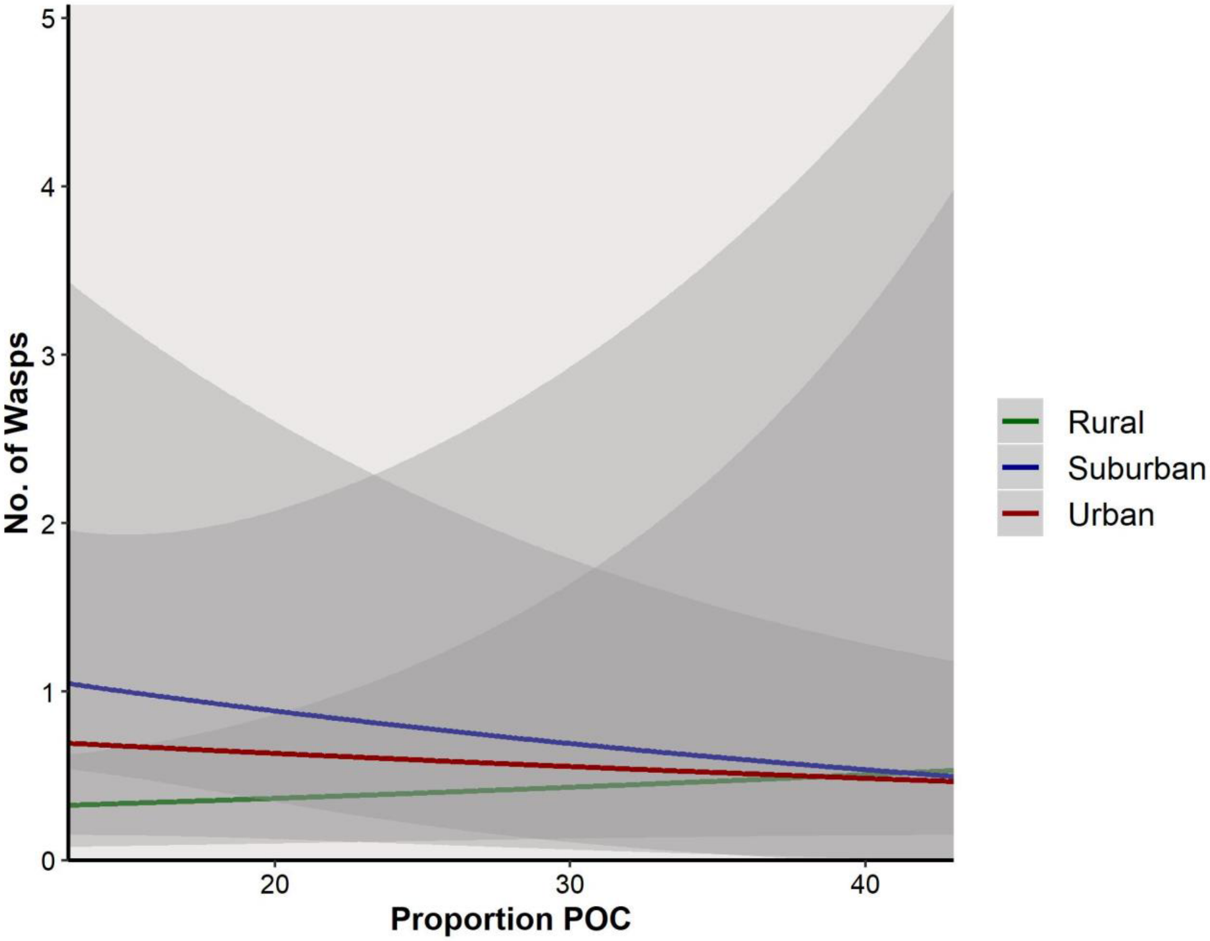

Figure 3.3. Interaction effects between the proportion of the population within a census tract made up of people of color and urbanization on the avereage number of wasps observed in Illinois gardens duing the summer of 2020. Rural sites are those with <20% impervious surface cover within a 750-m radius of the site, suburban sites are those with between 20% and 50% impervious surface cover, and rural sites are those with >50% impervious surface cover. Gray shaded areas represent 95% confidence intervals.

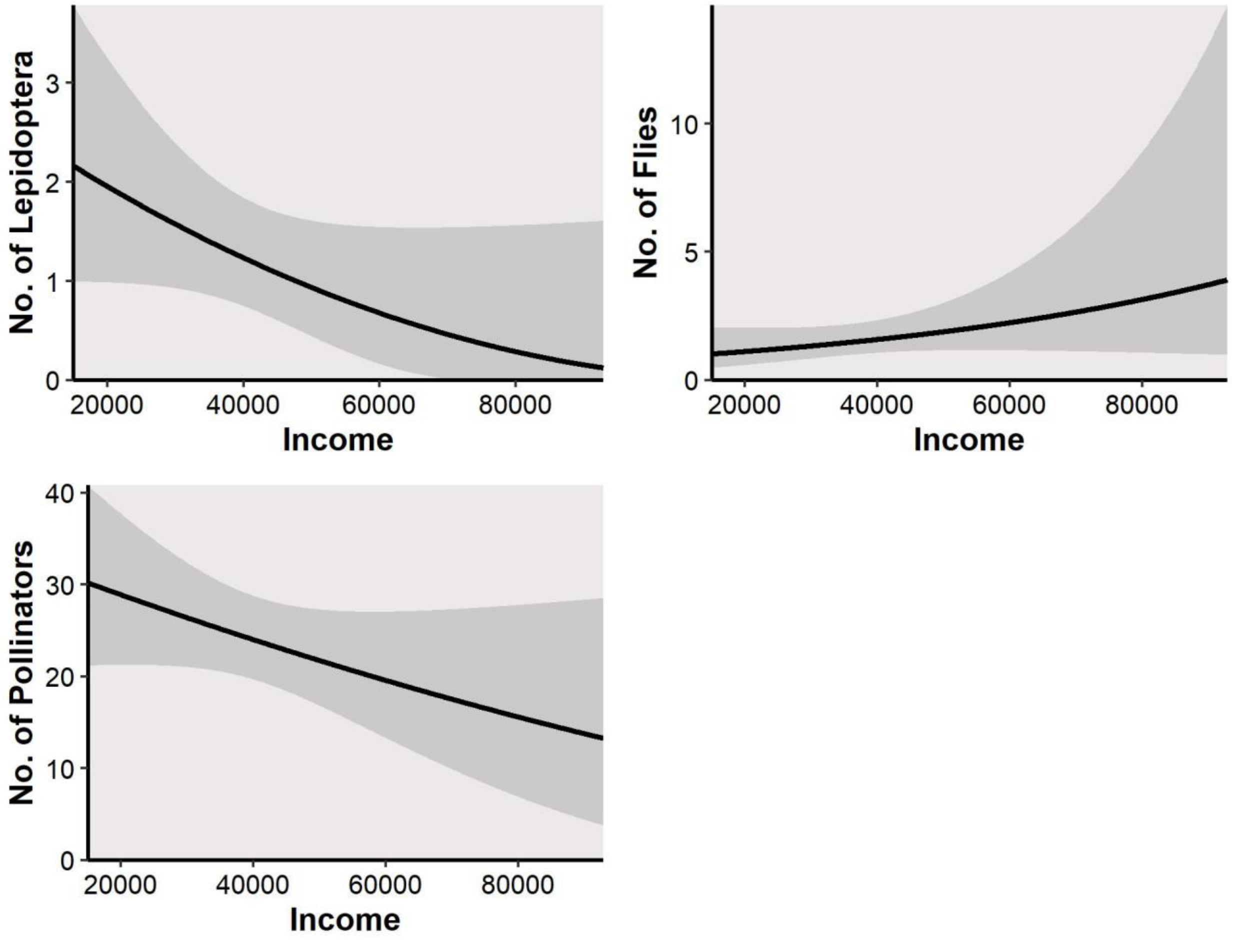

Figure 3.4. Relationships between average per capita income within a census tract and the average number of pollinators observed in Illinois gardens during the summer of 2020. Predicted values were calculated based on competitive models that included income as an independent variable. Gray shaded areas represent 95% confidence intervals.

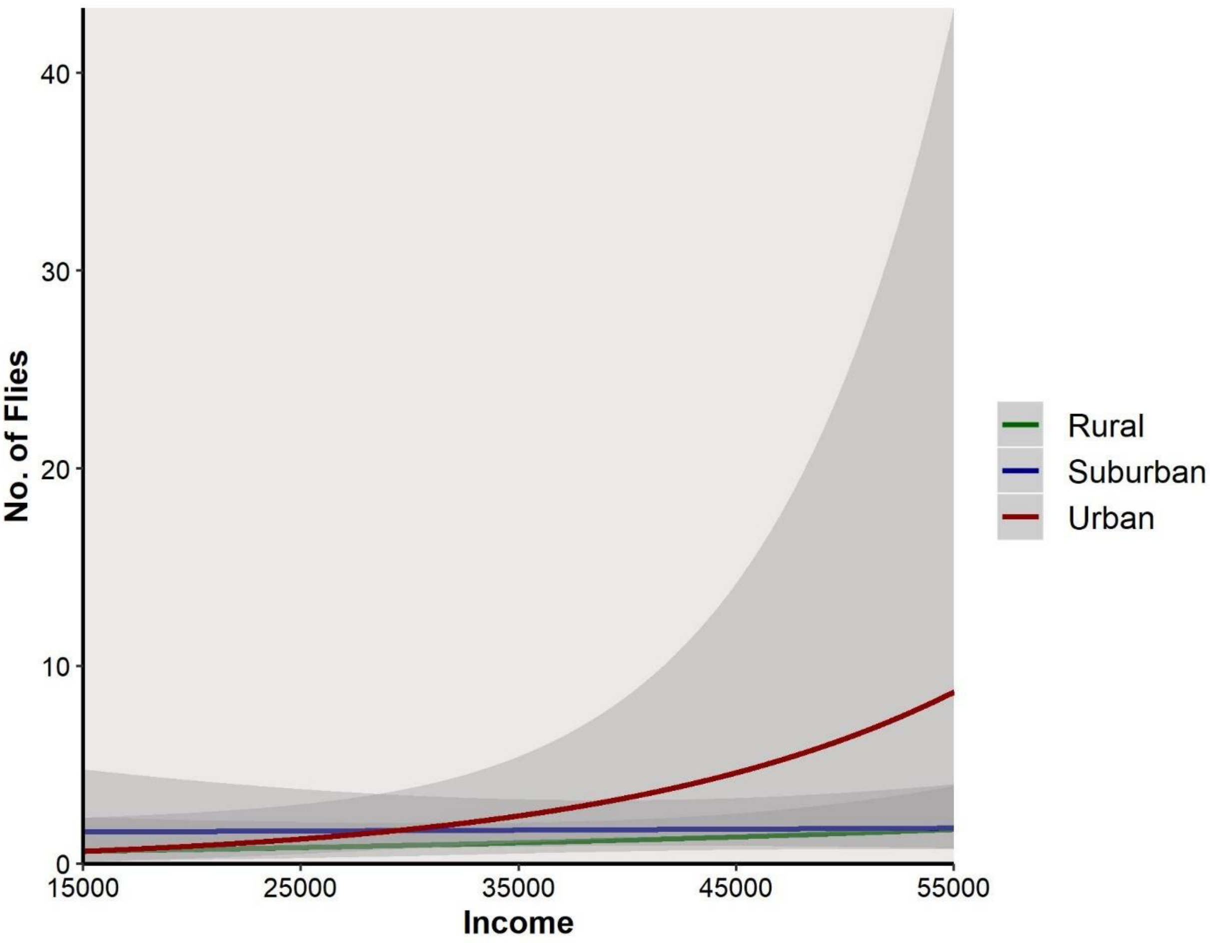

Figure 3.5. Interaction effects between average per capita income within a census tract and urbanization on the average number of flies observed in Illinois gardens duing the summer of 2020. Rural sites are those with <20% impervious surface cover within a 750-m radius of the site, suburban sites are those with between 20% and 50% impervious surface cover, and rural sites are those with >50% impervious surface cover. Gray shaded areas represent 95% confidence intervals.

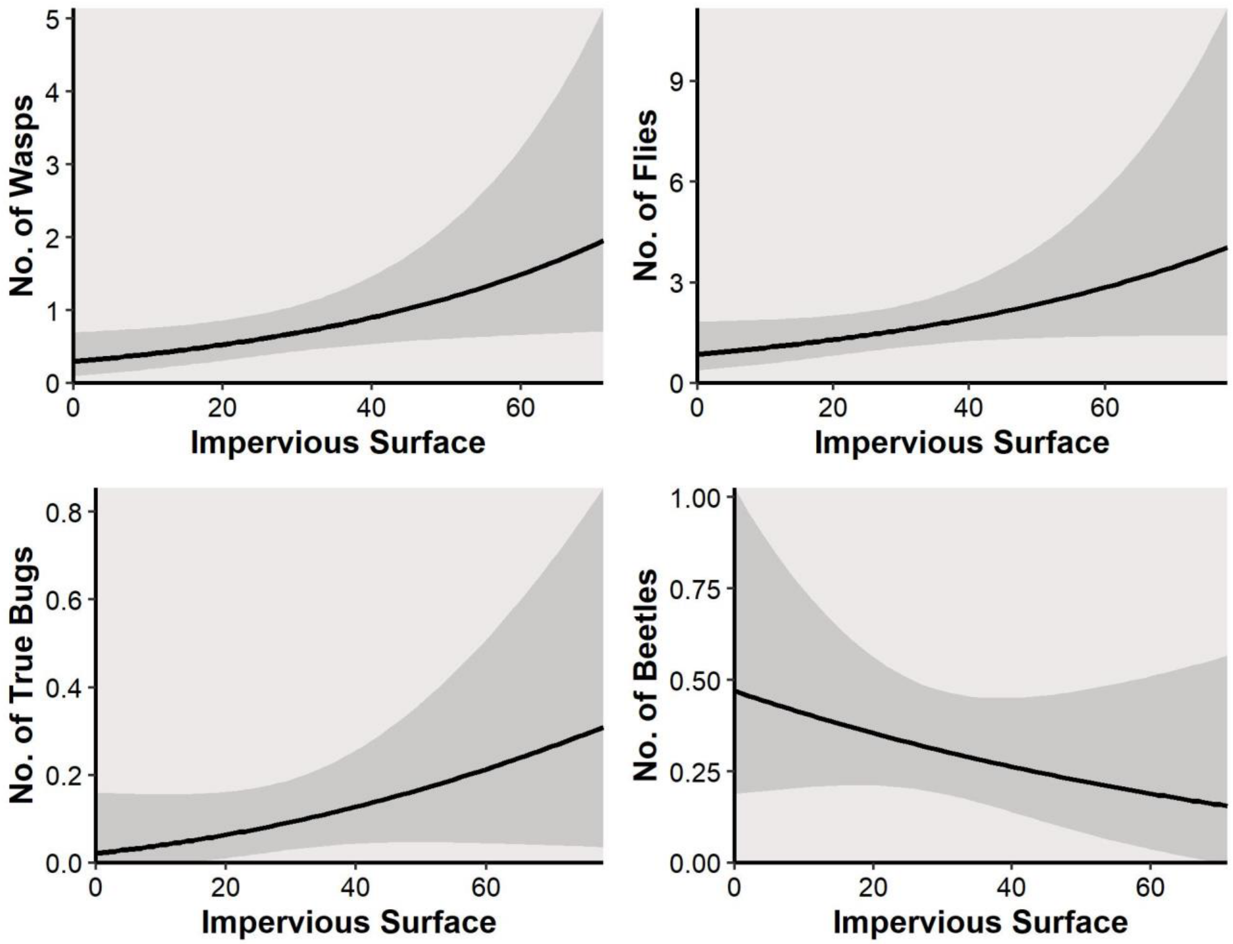

Figure 3.6. Relationships between impervious surface cover in the landscape surrounding study sites and the average number of pollinators observed in Illinois gardens during the summer of 2020. Impervious surface cover was measured within a 250-m radius for flies and true bugs, a 750-m radius for wasps, and a 1-km radius for beetles. Predicted values were calculated based on competitive models that included impervious surface cover as an independent variable. Gray shaded areas represent 95% confidence intervals.

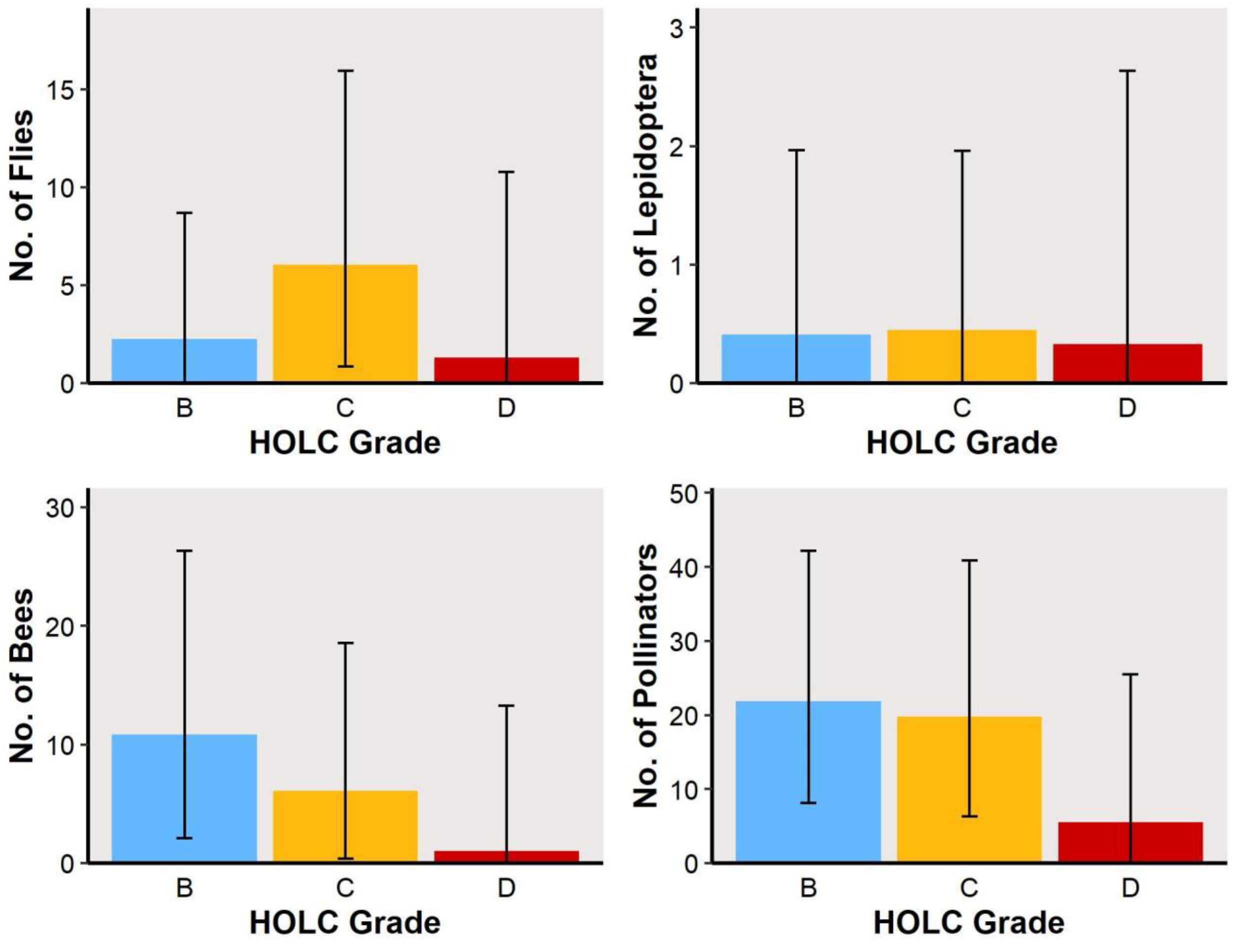

Figure 3.7. Relationships between historic housing discrimination practices and pollinator visitation. The average number of pollinators observed in Illinois gardens is shown at study sites located in historically redlined areas that were assigned grades of B, C, and D by the Home Owner's Loan Corporation (HOLC Grade). Error bars represent 95% confidence intervals. None of the relationships shown are statistically significant.

CHAPTER 4: ACCESSIBILITY OF PARTICIPATION IN A POLLINATOR-FOCUSED COMMUNITY SCIENCE PROJECT[3]

Abstract

Ecological research, education, and community engagement were interrupted globally in the spring and summer of 2020 due to the COVID-19 pandemic. To allow for continued data collection and to provide opportunities for people to interact with nature, we developed a community science project focusing on pollinator visitation to gardens and lawns. To evaluate the accessibility of this project to volunteers, we conducted 28 semi-structured interviews with people who participated in the project training. Interviewees experienced a number of barriers to participation, the most common of which were difficulty with the data collection procedure, challenges using technology, and lack of access to technology. These barriers were reported by 22, 11, and 9 participants, respectively. However, components of the online training and data collection procedure helped overcome some of these barriers. Strategies such as using a hybrid training format, simplifying the use of technology during training, and incorporating active learning into online workshops could increase participation in community science projects, both in the context of the pandemic and moving forward into a post-pandemic world.

Introduction

Ecological research, education, and community engagement were interrupted globally in the spring and summer of 2020 due to the COVID-19 pandemic (Marinono et al., 2020; Osafo, 2020;

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[3] L.R. Lynch and J.R. Miller. Prepared for submission to *Ecology and Evolution*

Radecki & Schonfeld, 2020). Research during this time was limited to ‘essential’ activities, and many ecologists canceled their field seasons (Radecki & Schonfeld, 2020). Additionally, both formal and informal in-person education was stalled for K-12 and university students as well as for adults (Marinono et al., 2020; Osafo, 2020). Outreach programs paused many activities and moved others online (Main et al., 2020; Osafo, 2020). The resulting lack of ecological research and education may be filled in part through the use of community science.

Community science refers to participation in research by people who are not professional scientists and is also frequently referred to as “citizen science” (Kovaka, 2021; Tulloch et al., 2013). In recent years, many organizations promoting public participation in scientific research have transitioned to the term “community science” to recognize the fact that participants come from diverse backgrounds and are not necessarily citizens of any particular country (Fuller, 2020; National Audubon Society, 2022). We use the term “academic scientists” to refer to researchers who are formally affiliated with a university (Eitzel et al., 2017).

Ecologists often use community science to increase the size of their datasets through crowdsourcing and to provide opportunities for public engagement and science education (Bonney et al., 2016). While the need for social distancing during the pandemic has caused some community science projects to pause activities (Escoto-Murillo & Alfaro, 2021; Lepenies & Zakari, 2021), others have been conducted fully online and seen steady or increased participation (Basile et al., 2021; Crimmins et al., 2021). Some projects that previously included in-person training have moved online to allow their continuation (Dwivedi, 2020; Smith & Hamed, 2020). Additionally, new projects have been developed to provide a source of data during lockdowns and create opportunities for students to engage in the scientific process remotely (Arbeláez-Cortés et al., 2021; Oberbauer et al., 2021).

Implementation of community science during the COVID-19 pandemic has been impacted by the ability of individual volunteers to participate. Inequities in the accessibility of participation in community science projects existed before the pandemic, particularly for groups historically underrepresented in science and those with lower incomes (Hobbs & White, 2012; Pandya, 2012). These inequities have been exacerbated by the pandemic. For example, people who do not have internet access and children who do not have a parent available at home to help them have been unable to participate in online community science projects (Smith & Hamed, 2020; Van Haeften et al., 2020).

Prior to the pandemic, we had been planning a field season in Chicago, IL USA, focusing on the visitation of pollinators to different types of green spaces along an urban-rural gradient. However, due to our inability to conduct socially-distanced fieldwork in an urban setting, our field season was canceled. Simultaneously, educational and volunteer opportunities offered by Illinois environmental organizations such as the Master Gardener and Master Naturalist programs (Illinois Extension, 2022a, 2022b), neighborhood gardening groups, and school environmental clubs were reduced or canceled entirely. We instead developed a community science project that would allow data collection during the 2020 field season and provide opportunities for the public to participate in a meaningful outdoor activity during the lockdown period. Our research interests were particularly well-suited to community science during the pandemic because volunteers could remain close to home while collecting data from their yards or neighborhood green spaces. Additionally, our research focused on insect pollinators, a charismatic functional group that is easily observable in urban and suburban settings. Here, we describe our project, evaluate its accessibility, and consider factors that impact participation in community science during the COVID-19 pandemic and moving into a post-pandemic world.

Methods

Community Science Project Design and Implementation

Project Goals

We had two overarching goals. First, we wanted to maintain data collection during the lockdown period that occurred in 2020. Our research interests were focused on understanding the impacts of urbanization on pollinator abundance and comparing pollinator visitation to different types of urban green spaces. Second, we wanted to provide opportunities for people to connect with nature and engage with the scientific process during the lockdown period.

Recruitment

Volunteers were recruited by contacting environmental and gardening organizations located in Illinois, USA. We contacted all Illinois Master Gardener and Master Naturalist groups in the state (Illinois Extension, 2022a, 2022b) as well as all gardening organizations in the greater Chicago area for which we could find publicly available contact information. Additionally, we contacted individuals, schools, and organizations with whom we had previously collaborated (Figure 4.1).

Training

All community scientists participated in a two-hour Zoom (Zoom Video Communications, 2022) training session before beginning data collection. We began the training with a short lecture reviewing the topic of global pollinator declines, proposing the idea that urban areas may play an important role in pollinator conservation, and sharing the results of our previous research focusing on urban pollinators. Next, we reviewed the identifying characteristics of thirteen broad groups of pollinators, using Zoom polls (Zoom Video Communications, 2022) to provide participants with opportunities to practice identification. Participants then worked in small groups in Zoom breakout rooms (Zoom Video Communications, 2022) to identify a series of specimen photos. We put these photos into a Google Doc (Google, 2022) and shared the link with participants. Participants were asked to open the link, review the photos together, and write what they thought was the correct pollinator group for each photo. After coming back together to review the correct answers in a large group and address points of confusion, participants were asked to go outside and complete a practice survey. This provided an opportunity for volunteers to gain confidence through practice and identify any concerns with the data collection procedure. Finally, participants returned to the Zoom room (Zoom Video Communications, 2022) to share their experiences observing pollinators and ask questions that occurred to them during their practice survey.

We held 12 training sessions attended by a total of 161 participants. Training sessions were conducted in May and June of 2020 during daylight hours to allow for outdoor practice. After each training session, attendees were emailed a copy of the presentation and the data collection sheet as well as a PDF of the “Upper Midwest Citizen Science Monitoring Guide for Native Bees” (Jordan et al., 2016).

Data Collection

Community scientists were instructed to select one or more study sites to monitor throughout the summer and fall. They could select an ornamental garden, a vegetable garden, a registered pollinator garden, or a lawn. If their selected garden or lawn was very large, they were asked to select a 152.4 m^2 (500 ft^2) within it to monitor. There was no minimum site area required for participation. Some participants opted to monitor two sites.

Community scientists were asked to conduct pollinator surveys at their selected site(s) at two-week intervals between June and November. During each survey, they spent 30 minutes walking slowly through their garden and recording all arthropods or birds that they observed landing on the reproductive parts of a flower. They were instructed to time themselves in a way that would allow them to observe the full site, and to spend the most time observing the most active flowers. If an individual pollinator was observed more than once during a survey, they were instructed only to record it the first time that they saw it (methods modified from Hülsmann et al., 2015; Jordan et al., 2016). Pollinators were identified to the same 13 groups presented during training.

Participants also recorded the site type (vegetable garden, ornamental garden, registered pollinator garden or lawn), plot size, and environmental data including temperature, wind speed, and cloud cover on each visit. We asked them to collect data between 0900 and 1700 h on sunny days when the wind speed was below 16.1 km/h (10 mph) since pollinators tend to be most active under these conditions. Finally, community scientists could either submit their observations through an online Qualtrics survey (Qualtrics, 2020) or mail us a hard copy of their data collection sheet.

Evaluation of Project Accessibility

To evaluate the accessibility of this project, we conducted 28 semi-structured interviews with people who attended project training sessions. Our goal was to obtain a sample size that would allow us to reach “saturation,” or a point at which conducting additional interviews does not produce new information (Guest et al., 2006). Saturation is generally reached within 20 interviews in qualitative studies using semi-structured interviews and having a relatively homogenous population (Guest et al., 2006; Morgan et al., 2002; Namey et al., 2016). Our results do confirm that we were able to reach saturation. Of the 130 main ideas that we identified in our interview transcripts, only 3% first appeared in the last three interviews that we conducted (Guest et al., 2006).

To recruit interviewees, we emailed an invitation to all people who attended a project training session. Two months after the invitation was sent, we began to send follow-up emails to randomly selected groups of 20 training attendees. We sent follow-up emails to one group of attendees every other week until we had scheduled 25 interviews.

Interview questions focused on engagement in community science during the COVID-19 pandemic, barriers to participation in our project, access to and use of required technology, and recommendations for improving project accessibility (Appendix 4). We were interested in understanding barriers resulting from the pandemic as well as barriers that are likely to continue to impact participants post-pandemic. Interviews were conducted by Zoom or by phone between 24 September 2021 and 17 January 2022. All interviews were recorded and transcribed.

Data analysis comprised several rounds of qualitative coding (Saldaña, 2021). In our first two rounds, we divided the information from the interviews into categories based on the topic being discussed (e.g., technology use, access to green space, data collection). The third round involved the use of in-vivo and process coding to explore the content of the interviews more deeply. In-vivo codes are words or short phrases used by the interviewees while process codes are gerund verbs describing the activity of interviewees. In-vivo and process codes were developed a posteriori based on the content present in the interview data (Saldaña, 2021). Finally, in our fourth round, we consolidated the codes from round three to identify a list of all barriers to participation, factors that facilitated participation, and suggestions for improving accessibility that were identified by interviewees.

Results

Barriers to Participation

Community scientists identified a number of barriers to participation in this project (Table 4.1). The most frequently reported challenge, noted by 22 participants, was a difficulty identifying pollinators in the field. Interviewees explained that many pollinators are quite small and move quickly. Some also found that pollinators from the different groups could sometimes look very similar and that their lack of experience with insect identification may have contributed to their difficulty. In addition to problems with identification, several interviewees noted that it could be challenging to get an accurate count of pollinators present at their study sites. They explained that it was sometimes difficult to count pollinators when they were particularly abundant and that at times it was hard to know whether a pollinator that they observed had newly arrived or had already been recorded at a different flower.

Many community scientists overcame difficulties with data collection by referring to training materials, insect ID books, or information available online. One common strategy was to take photos of unknown pollinators during data collection and confirm the identification afterwards. Participants often reported talking to each other or emailing an academic scientist involved in the project for clarification. Despite the use of these tools, problems with insect identification did appear to present a real barrier to participation in the project. One interviewee explained that she almost decided not to submit her data due to concerns about accuracy saying, "Because I wasn't exactly sure what I was seeing, I did not want to send in my paperwork… It just kind of stopped my participation."

Difficulty using technology and a lack of access to technology were the second and third most common factors identified as barriers to participation in the project, reported by 11 and nine participants, respectively. The most frequently reported access issue was problems with internet connectivity. Regarding technology use, some participants did not know how to use Zoom or Google Docs (Google, 2022; Zoom Video Communications, 2022), experienced technological malfunctions during the training, or simply did not enjoy the time spent at the computer. Additionally, several had trouble with the online data submission portal or had difficulty taking clear photos for insect identification. While no interviewees reported that a lack of access to technology or technological difficulties prevented them from participating, it was clear that issues related to technology did impact the quality of the experience for some. For example, a long lag time resulting from a poor internet connection made it difficult for one interviewee to participate in small group discussions while another was unable to ask questions because she did not know how to use the Zoom chat feature (Zoom Video Communications, 2022).

For five participants, issues related to health and physical disabilities presented an additional barrier. While no interviewees reported that contracting COVID impacted their participation, one community scientist said that a family member died of COVID during the project period and that as a result, she took on additional family responsibilities that limited her ability to participate. Mental health concerns were also cited as a barrier. For example, one interviewee noted that stress caused by living through the pandemic impacted her participation in the project explaining, “the pandemic was such a weird time and I think we all felt so shut down and dysfunctional that the thought of doing anything just seemed like a lot of work.”

Limited access to high quality pollinator habitat and low levels of pollinator activity were identified as barriers by four participants. While all interviewees reported having access to either a garden or a lawn in which to collect data, several noted that data collection was less fun at sites with lower levels of pollinator activity. While they were aware that the data they collected were still valuable despite the low pollinator abundance, some people felt that the quality of their experience was reduced.

Finally, five interviewees reported that it was difficult to find the time to participate in this project due to responsibilities including work and childcare.

Factors Facilitating Participation

Interviewees identified a number of factors that helped them overcome barriers to participation (Table 4.2). The most frequently mentioned factor facilitating participation, noted by 20 interviewees, was the project training. Several confirmed that the content covered provided them with the knowledge and skills that they needed to participate in the project and said that the materials provided were useful references during data collection. Additionally, participants noted that training components including question-and-answer periods, practice in small groups, and the use of Zoom polls (Zoom Video Communications, 2022) to check understanding increased their confidence with pollinator identification.

The structure of the data collection and submission procedures were also identified as factors that made it easier to participate in the project by 17 and ten participants, respectively. Interviewees noted that the data submission procedure was straightforward and flexible and appreciated that they had the option to either submit the data online or send it by mail. Participants noted that the data collection procedure was straightforward, easy to learn, and did not require prior expertise. Some found that the time commitment was reasonable and appreciated the flexibility of being able to collect data any time between 900 and 1700 h on any day with the appropriate weather conditions.

Eight participants noted that collaborating with each other and with academic scientists helped them overcome barriers related to pollinator identification. Community scientists involved in the project created a Facebook group that they used to share photos and ask questions. Some participated in the project with a friend and helped each other via phone or email. Additionally, community scientists could email academic scientists to ask questions or confirm identifications and several optional mid-summer question-and-answer periods provided opportunities for participants to clarify points of confusion.

Participants noted that in some ways, the online format of the training made it even easier for them to be involved in the project than it would have been under normal circumstances. Twelve interviewees explained that it was convenient not to have to drive to the training. They saved time and avoided unsafe road conditions. For some, not needing to drive also improved the quality of the experience. For example, one person noted that “the nice thing about the online training is being able to connect with people from across the state, not just my local group.”

Participants were also aware that personal circumstances made it easier for them to be involved in the project. Four and three interviewees, respectively, noted that access to technology and pollinator habitat facilitated their participation. Additionally, seven said that prior experience with technology was helpful. The shared experience of the pandemic contributed to many participants’ familiarity with technology. Many interviewees were already accustomed to using Zoom (Zoom Video Communications, 2022). One explained, “That’s COVID bread. Everything I’ve been doing has been by Zoom.”

Discussion

Many of the barriers encountered by community scientists participating in our project were caused or exacerbated by COVID-19 while others were present before 2020 and will likely continue to impact engagement regardless of the pandemic’s progression. Many technological barriers occurred due to the online format of the training. Additionally, interviews suggest that difficulties with the data collection procedure were exacerbated by the need for social distancing. Several interviewees noted that they would have felt more comfortable with pollinator identification if they had been able to observe physical specimens during the training. Similarly, the pandemic reduced the free time that people had to participate in the project by creating additional caretaking and homeschooling responsibilities for some and led to a lack of access to pollinator habitat by preventing people from traveling for data collection. Stress caused by COVID-19 also created or exacerbated mental health challenges that impacted participation.

Most barriers that impacted participants in our project would have been present to some extent even if COVID-19 had not occurred. Difficulties with the data submission portal would have been present regardless, and participants likely would still have found it difficult to identify pollinators in the field. Busyness, lack of access to green space, and health problems would have impacted some participants under any circumstances.

Barriers to participation in community science do not impact all people equally. People belonging to minoritized groups often live farther away from green space (Saporito & Casey, 2015; Schell et al., 2020), are required to work unpredictable hours including mandatory overtime (Danziger & Boots, 2008), and have less access to high quality health care (Orentlicher, 2018), all of which are factors that exacerbate barriers. New challenges brought about by the pandemic also disproportionately impact minoritized groups. For example, people of color and people from lower-income households were nearly twice as likely to experience internet connectivity issues when attempting to access educational material during the first year of the pandemic (Means et al., 2021). Inequities occurred in our project as well. Although we did not collect demographic information, we did observe that, on average, data was collected from wealthier and whiter neighborhoods than the state average (US Census Bureau, 2019). Additionally, many participants were retired, and younger people noted that work and parenting made involvement more challenging.

Although COVID-19 exacerbated many barriers to involvement in our project, it also increased accessibility for some interviewees by allowing them to participate from home. This demonstrates a tradeoff in which some community scientists are negatively impacted by the transition to online training while others are positively impacted.

As several interviewees noted, one way to maximize the accessibility of participation in community science projects could be the use of a hybrid training format. The training could be organized in a way that allows participants to attend either in-person or via Zoom (Zoom Video Communications, 2022) depending on their preference. Hybrid learning has become more common as the development of vaccines and treatments for COVID-19 have reduced the severity of health risks associated with in-person events. Research suggests that learners in hybrid settings build social relationships with teachers and classmates, feel they have more control over their learning, and achieve similar test scores as learners in fully in-person settings (Raes et al., 2019). A hybrid format would allow people who face technological barriers to participate in-person while allowing those who face transportation barriers to participate online.

In cases where the continued use of the online format is preferred, technological barriers could be reduced by simplifying technology use during the training. For example, activities using breakout rooms could be traded for activities that can be conducted with the full group. The downside to this is that it would likely lead to less use of active learning. Since active learning improves student performance and disproportionately benefits learners belonging to minoritized groups, this change could lead to community scientists feeling less prepared to collect data after completing the training (Freeman et al., 2014; Haak et al., 2011). Another option would be to use an asynchronous format for the training since this is often more accessible to learners with limited internet connectivity (Means et al., 2021). However, asynchronous learning interferes with cooperation and can negatively impact cognitive processes and learning outcomes (Guo, 2020; Peterson et al., 2018).

Incorporating active learning strategies into online training may help prevent difficulties with the data collection procedure. Although it can be more challenging to incorporate active learning strategies online, doing so may improve understanding of content, critical thinking skills, motivation, and enjoyment (Means et al., 2021; Nguyen et al., 2021; Rossi et al., 2021). In this project, we used Zoom polls, breakout rooms, and individual outdoor practice to incorporate active learning into our online training.

Simplifying the pollinator identification categories that community scientists are asked to use was recommended by two interviewees and may also make data collection easier. While there is evidence that community scientists can detect trends in pollinator abundance and accurately identify pollinators to broad taxonomic groups, it is common for community scientists to struggle when they are asked to identify pollinators to taxonomic levels narrower than order (Kremen et al., 2011; Roy et al., 2016). Using a smaller set of identification categories would likely have reduced interviewees' confusion and increased their confidence during data collection.

Barriers to participation in community science that we have identified through this study occurred in the context of the COVID-19 pandemic but are likely to be relevant in a post-pandemic world as well. While technology use was particularly essential during the pandemic due to the need for social distancing, online training and data submission were frequently components of community science projects prior to the pandemic and will continue to be used in years to come. Participation in community science has been shown to provide a number of benefits including opportunities for learning about ecological topics, the development of a stronger sense of place, and an increased interest in and identity with topics related to science and conservation (Bonney et al., 2009; Hiller & Kitsantas, 2014). Implementing changes to address barriers to participation will provide opportunities for more people to engage in these valuable experiences.

Figure and Tables

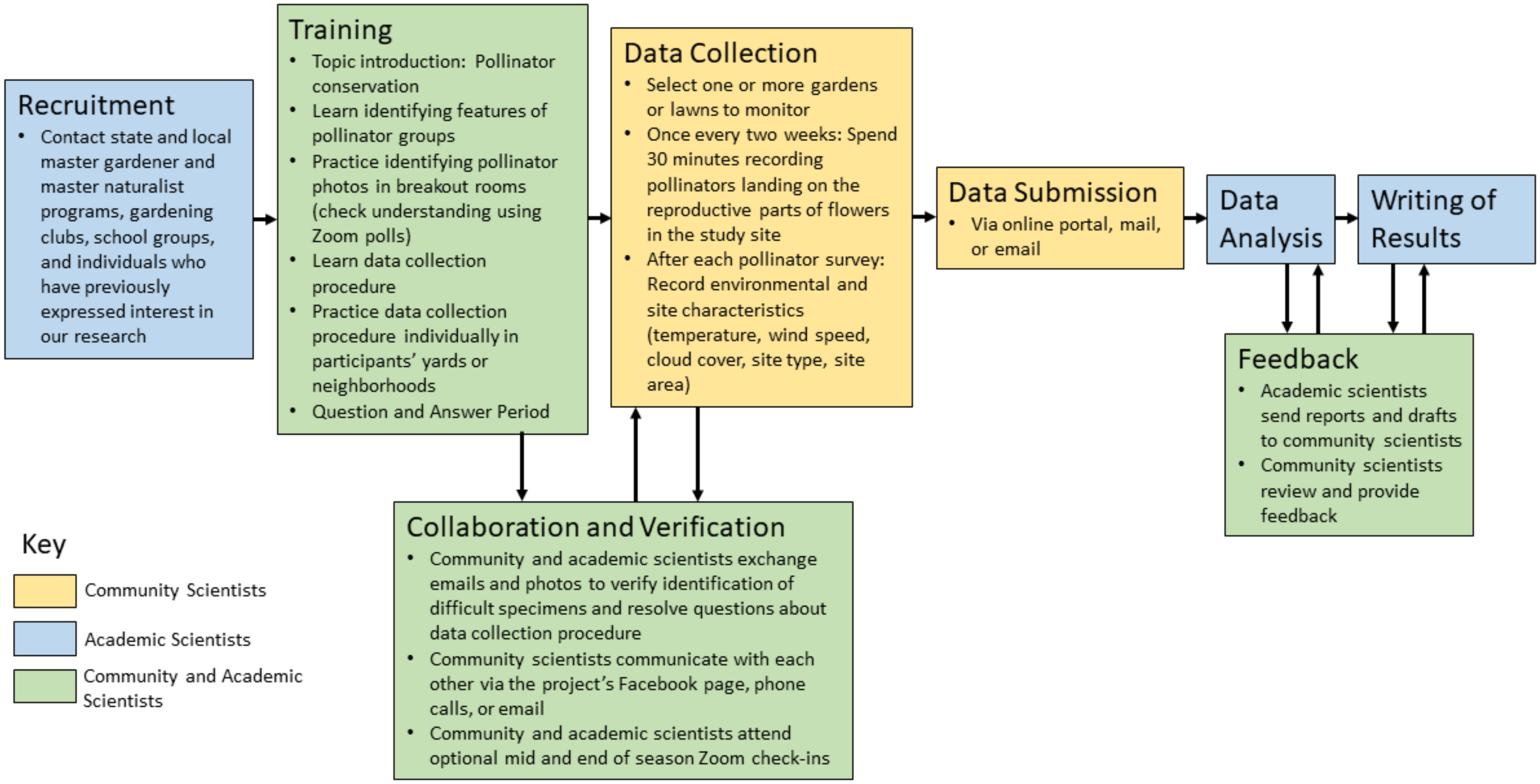

Figure 4.1. Diagram describing the process that we used to implement a pollinator-focused community science project beginning in the first summer of the COVID-19 pandemic.

Table 4.1. Barriers to Participation in a Pollinator-Focused Community Science Project Occurring during the COVID-19 Pandemic

Barrier	Participants Experiencing Barrier	Explanation
Difficulty with Data Collection Procedure	**22**	
Pollinator Identification	21	Difficulty identifying pollinators during data collection. Participants noted a number of factors that made identification difficult including difficulty seeing insects that were small and/or moving quickly, difficulty remembering the identification groups, a lack of prior knowledge or experience, and the fact that sometimes two different species can look very similar.
Pollinator Counts	5	Difficulty getting an accurate count of pollinators when they were particularly abundant, or not knowing whether an individual pollinator had already been counted.
Site Selection	1	Difficulty measuring a potential study site and deciding which flower patches within a garden to include.
Understanding Instructions	1	Confusion about the level of detail needed in pollinator identification.
Use of Technology	**11**	
Data Submission	6	Finding data submission to be difficult due to technological problems or not knowing how to use the online form or finding data submission to be time consuming.
Participation in Training	6	Finding participation in the online training to be difficult due to problems using Zoom or Google Docs, a lack of confidence using technology, or not enjoying training conducted in an online format
Difficulty Using Camera	3	Difficulty getting photos of pollinators that are clear enough to be usable for identification purposes.
Access to Technology	**9**	
Access to Internet or Computer	8	Not having Wi-Fi at home, having problems with internet connectivity that interfered with participation in the training, or having an older device that made participation more difficult
Access to Printer	1	Not having access to a color printer that could be used to print the insect identification materials that were provided electronically.
Access to Camera	1	Not having access to a camera that can take high-quality photos.

Table 4.1 cont.

Time to Participate	**5**	Not having time to participate in the project for reasons including working or taking on extra responsibilities because of the COVID pandemic (e.g., childcare).
Health and Physical Disability	**5**	
Physical Health	3	An injury or illness that prevented participation in the project. No participants said that getting COVID themselves prevented them from participating.
Mental Health	2	Difficulty participating due to emotional strain caused by the COVID pandemic.
Mobility	1	Difficulty accessing study sites due to a physical disability.
Vision	1	Difficulty seeing pollinators due to vision loss.
Access to Pollinators	**4**	Having limited access to high quality garden areas in which to monitor pollinators, or low pollinator activity in nearby green space leading to boredom during data collection.
Weather	**2**	Weather conditions that either made it unpleasant to be outside (e.g., heat) or that were outside of the conditions required for data collection (e.g., rain, full cloud cover).
Not Kid-Friendly	**1**	Children not wanting or being able to participate, and parents having difficulty participating because their children do not want to participate with them.
Pet Interference	**1**	Difficulty with data collection due to a dog chasing pollinators away from the plants.
Public Spaces Crowded	**1**	Not wanting to collect data in crowded public spaces, in particular after the first few months of social distancing when people resumed spending more time outside of their homes.

Table 4.2. Factors Facilitating Participation in a Pollinator-Focused Community Science Project Occurring during the COVID-19 Pandemic

Facilitating Factor	Number of Participants Saying This Helped Them	Explanation
Training Aspects	**20**	
Reference Materials	11	Participants were sent PowerPoint slides from the training including information about pollinator identification.
Group Work in Breakout Rooms	8	Participants practiced identifying photos of pollinators in small groups in Zoom breakout rooms.
Content Covered	5	Some participants noted that the information covered during the training was helpful for them during data collection. In particular, some people appreciated the information about pollinator identification that was covered.
Zoom Polls	3	During the overview of key characteristics to be used for pollinator identification, periodic Zoom polls were used to check understanding and provide an opportunity for practice.
Time for Questions	2	Time was set aside for question and answer periods at several points during the training.
Hands-On Practice	1	Participants were given time during the training to go outside and practice the data collection procedure in their yards.
Clarity of Instruction	1	One participant noted that the instructions for the data collection procedure were clear and easy to follow.
Data Collection Procedure	**17**	
No Travel Required	12	Participants saved time and avoided unpleasant weather or road conditions by being able to participate in the training from home.
Data Collection Easy	5	The data collection procedure is easy to learn and does not require prior expertise.
Time Commitment Small	4	Participants are asked to collect data once every two weeks over the course of the summer and fall.

Table 4.2 cont.

Timing Flexible	3	There is not a set day and time at which data must be collected. Participants can choose the day and time that works best for them.
Study Site Flexible	1	The study site can be any garden or lawn and there is no minimum study site size. Participants are able to collect data from their own yards.
Data Submission Procedure	**10**	Some participants noted that they found the data submission procedure to be simple and straightforward. There is also flexibility in that participants can either submit their data through an online portal or send it by mail.
Data Collection Support	**8**	
Collaboration between Community Scientists	4	Community scientists helped each other identify specimens by sharing photos in a group chat that they created. Some participants signed up for the project with a spouse or friend and helped each other throughout the project.
Collaboration with Academic Scientists	6	Academic scientists were available to answer questions by email. Additionally, an optional question and answer session was held mid-summer.
Technology Use	**8**	
Experience Using Technology	7	Some community scientists noted that their prior experience using computers and the internet as well as specific programs such as Zoom and Google Docs made it easy for them to participate in this project. Several community scientists noted that as a result of the pandemic, they had already learned how to use Zoom before they participated in my project.
Enjoy Using Technology	1	One community scientist noted that her enjoyment of technology has made it easy for her to keep up with new developments and programs, which helped her during this training.
Technology Easy to Use	1	One community scientist thought that Zoom is a relatively easy program to learn.

Table 4.2 cont.

Access to Technology	**4**	
Access to Camera	2	Some community scientists said that having access to a nice camera that allowed them to take pictures of pollinators to help with identification was helpful.
Access to Computer or Internet	2	Some community scientists said they were aware that their access to the computer and the internet made it easy for them to participate in this project.
Access to Green Space	**3**	Community scientists noted that having access to green space allowed them to easily participate in this project. Some noted their access to native plants and high quality pollinator habitat.
Ecological Factors	**3**	
Pollinator Abundance/Diversity Low	3	A few community scientists said that they tended to see a few common species at their study sites, which made identification easier.
Slow Pollinators	1	One community scientist said that they pollinators didn't move as fast as some other insects, which made it easier to get a look at them.
Experience with Ecology	**2**	Community scientists noted that their prior experience with pollinators or with other ecological projects helped prepare them to participate in this project.
Appreciation for the Outdoors	**1**	One community scientist said that her enjoyment of time spent in nature provided motivation to participate in the project.
Retirement	**1**	One community scientist recognized that being retired allowed her to be available at the times of day that pollinators tend to be most active.

CHAPTER 5: CONCLUSIONS AND IMPLICATIONS

The essential ecosystem services that pollinators provide are under threat due to global declines in populations of bees and butterflies (Goulson et al., 2015; Potts et al., 2010a; Warren et al., 2021). Additionally, the ability of many people to receive these benefits is further limited by environmental injustices and barriers to participation in environmental education and volunteering opportunities. My research identifies racial inequities in the abundance of pollinators visiting home gardens along an urban-rural gradient and finds that a number of barriers, including new challenges resulting from the COVID-19 pandemic as well as enduring issues likely to extend far beyond the resolution of this global health crisis, limit participation in pollinator-focused community science.

Given the many processes limiting human access to benefits provided by pollinators, a multi-pronged approach is needed to address this issue. First, we must continue to direct research and management efforts towards the goal of pollinator conservation. Strategies that enhance the availability of floral resources are likely to be especially beneficial (Goulson et al., 2015). My research suggests that initiatives such as pollinator garden registration programs that encourage community members to provide small patches of floral resources may be beneficial while other studies have demonstrated the value of enhancing floral resources on agricultural land and maintaining semi-natural habitat (e.g., Carvell et al., 2006; Pywell et al., 2006; Rundlöf et al., 2008). Providing nesting resources and larval host plants, limiting the use of pesticides, and avoiding the introduction of new parasites and diseases are also likely to benefit pollinator populations (Goulson et al., 2015; MacIvor, 2016; Thogmartin et al., 2017).

We may also be able to increase the opportunities that people have to interact with pollinators by removing barriers to participation in environmental education and volunteering. This is particularly necessary given challenges that have been caused or exacerbated by the COVID-19 pandemic. In Chapter 4, I suggest that using a hybrid training format that allows learners to choose whether to participate online or in-person provides flexibility that may maximize accessibility. Keeping technology simple and making use of active learning strategies may also improve accessibility as well as learning outcomes (Means et al., 2021; Nguyen et al., 2021; Rossi et al., 2021).

Finally, we need to work to correct inequities in access to nature. This may involve directing funding for pollinator conservation towards less wealthy and less white neighborhoods. My research indicates that pollinator abundance is lower in neighborhoods where people of color live while other studies have identified inequities based on income (Baldock et al., 2019; MacIvor, 2015). Additionally, working to address the root causes of environmental injustice, such as racism and classism, may help solve this problem. For example, addressing racism may help to correct issues like housing discrimination that are thought to contribute to inequities in access to biodiversity (Schell et al., 2020). While overt forms of housing discrimination such as redlining are no longer legal, more discrete practices like “racial steering” in which real estate agents encourage people of color to buy homes in neighborhoods with lower housing quality, continue to occur and may also impact biodiversity (Zonta, 2019).

The implementation of these strategies is particularly necessary in urban areas where residents often rely on gardening as a supplemental source of food and income and where access to biodiversity and green space is often limited. By addressing the multiple factors that limit opportunities to interact with pollinators, we may be able to extend the benefits that they provide to a wider range of people. Doing so has the potential to improve human well-being in cities, suburbs, and rural towns around the world.

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APPENDIX A: AIC_C TABLES (CHAPTER 2)

Table A.1. Results from first round of AIC_C used to select covariates for all models with honey bee abundance (sampled by survey) as the dependent variable

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Day	4	0.00	0.99	0.99	-625.26
Year	4	21.05	0.00	1.00	-635.79
Temperature	4	22.50	0.00	1.00	-636.51
Null	3	22.89	0.00	1.00	-637.73
Wind	4	23.49	0.00	1.00	-637.01
Cloud Cover	4	23.98	0.00	1.00	-637.25
Site Area	4	24.20	0.00	1.00	-637.36
Minute	4	24.88	0.00	1.00	-637.71

Table A.2. Results from first round of AIC_C used to select covariates for all models with bumble bee abundance (sampled by survey) as the dependent variable

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Day	4	0.00	1.00	1.00	-467.95
Site Area	4	35.77	0.00	1.00	-485.84
Null	3	38.07	0.00	1.00	-488.01
Cloud Cover	4	38.38	0.00	1.00	-487.14
Wind	4	39.04	0.00	1.00	-487.47
Year	4	39.34	0.00	1.00	-487.62
Temperature	4	40.07	0.00	1.00	-487.98
Minute	4	40.07	0.00	1.00	-487.99

Table A.3. Results from first round of AIC_C used to select covariates for all models with bumble bee richness (sampled by survey) as the dependent variable

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Day	4	0.00	1.00	1.00	-274.72
Site Area	4	23.72	0.00	1.00	-286.58
Cloud Cover	4	25.91	0.00	1.00	-287.68
Null	3	26.02	0.00	1.00	-288.76
Year	4	27.61	0.00	1.00	-288.53
Wind	4	28.04	0.00	1.00	-288.74
Temperature	4	28.05	0.00	1.00	-288.75
Minute	4	28.06	0.00	1.00	-288.75

Table A.4. Results from first round of AIC_C used to select covariates for all models with total bee abundance (sampled by survey) as the dependent variable

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Day	4	0.00	1.00	1.00	-658.03
Year	4	66.70	0.00	1.00	-691.38
Temperature	4	74.06	0.00	1.00	-695.06
Cloud Cover	4	82.84	0.00	1.00	-699.45
Null	3	86.86	0.00	1.00	-702.49
Site Area	4	87.01	0.00	1.00	-701.53
Wind	4	87.85	0.00	1.00	-701.95
Minute	4	88.26	0.00	1.00	-702.16

Table A.5. Results from first round of AIC$_C$ used to select covariates for all models with total bee abundance (sampled by bee bowls) as the dependent variable

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Year	4	0.00	1.00	1.00	-507.10
Day	4	33.54	0.00	1.00	-523.87
Minute	4	37.44	0.00	1.00	-525.82
Temperature	4	42.21	0.00	1.00	-528.21
Wind	4	42.37	0.00	1.00	-528.28
Cloud Cover	4	42.79	0.00	1.00	-528.49
Site Area	4	42.83	0.00	1.00	-528.52
Null	3	45.29	0.00	1.00	-530.77

Table A.6. Results from first round of AIC$_C$ used to select covariates for all models with total bee richness (sampled by bee bowls) as the dependent variable

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Year	4	0.00	1.00	1.00	-385.75
Day	4	45.29	0.00	1.00	-408.40
Minute	4	49.03	0.00	1.00	-410.27
Wind	4	50.51	0.00	1.00	-411.01
Temperature	4	50.06	0.00	1.00	-411.28
Cloud Cover	4	51.84	0.00	1.00	-411.68
Site Area	4	52.07	0.00	1.00	-411.79
Null	3	53.07	0.00	1.00	-413.31

Table A.7. Results from first round of AIC$_C$ used to select covariates for all models with flower density as the dependent variable

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Site Area	4	0.00	0.58	0.58	-1200.82
Null	3	3.06	0.13	0.71	-1203.37
Day	4	4.77	0.05	0.76	-1203.20
Wind	4	4.87	0.05	0.81	-1203.25
Year	4	4.89	0.05	0.86	-1203.26
Minute	4	5.00	0.05	0.91	-1203.32
Temperature	4	5.08	0.05	0.95	-1203.36
Cloud Cover	4	5.10	0.05	1.00	-1203.37

Table A.8. Results from first round of AIC$_C$ used to select covariates for all models with flower richness as the dependent variable

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Day	4	0.00	1.00	1.00	568.61
Temperature	4	57.52	0.00	1.00	539.85
Site Area	4	62.20	0.00	1.00	537.52
Null	3	73.78	0.00	1.00	530.70
Cloud Cover	4	74.71	0.00	1.00	531.26
Wind	4	74.83	0.00	1.00	531.20
Year	4	75.44	0.00	1.00	530.89
Minute	4	75.71	0.00	1.00	530.76

Table A.9. AIC$_C$ results from models describing the relationships between flower density, urbanization, and honey bee abundance sampled by surveys.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Flower Density+Urbanization	6	0	0.54	0.54	-616.12
Flower Density*Urbanization	7	0.59	0.40	0.95	-615.36
Flower Density	5	4.63	0.05	0.99	-619.46
Null	4	14.17	0.00	1.00	-625.26

Table A.10. AIC$_C$ results from models describing the relationships between flower density, urbanization, and bumble bee abundance sampled by surveys.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Flower Density	5	0.00	0.41	0.41	-466.58
Null	4	0.67	0.29	0.70	-467.95
Flower Density+Urbanization	6	1.35	0.21	0.91	-466.22
Flower Density*Urbanization	7	3.06	0.09	1.00	-466.03

Table A.11. AIC$_C$ results from models describing the relationships between flower density, urbanization, and bumble bee richness sampled by surveys.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Flower Density	5	0.00	0.43	0.43	-271.14
Flower Density+Urbanization	6	0.76	0.30	0.73	-270.49
Flower Density*Urbanization	7	1.18	0.24	0.97	-269.65
Null	4	5.10	0.03	1.00	-274.72

Table A.12. AIC$_C$ results from models describing the relationships between flower density, urbanization, and total bee abundance sampled by surveys.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Flower Density	5	0.00	0.55	0.55	-655.46
Flower Density+Urbanization	6	1.84	0.22	0.76	-655.32
Flower Density*Urbanization	7	3.01	0.12	0.88	-654.88
Null	4	3.08	0.12	1.00	-658.03

Table A.13. AIC$_C$ results from models describing the relationships between flower density, urbanization, and total bee abundance sampled by bee bowls.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Flower Density*Urbanization	7	0.00	0.87	0.87	-500.06
Flower Density	5	5.43	0.06	0.93	-504.85
Flower Density+Urbanization	6	5.53	0.05	0.98	-503.87
Null	4	7.88	0.02	1.00	-507.10

Table A.14. AIC$_C$ results from models describing the relationships between flower density, urbanization, and total bee richness sampled by bee bowls.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Flower Density*Urbanization	7	0.00	0.43	0.43	-382.02
Null	4	1.26	0.23	0.66	-385.75
Flower Density	5	1.87	0.17	0.83	-385.03
Flower Density+Urbanization	6	1.89	0.17	1.00	-384.01

Table A.15. AIC_C results from models describing the relationships between flower richness, urbanization, and honey bee abundance sampled by surveys.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Flower Richness*Urbanization	7	0.00	0.49	0.49	-616.49
Flower Richness+Urbanization	6	0.23	0.44	0.93	-617.65
Flower Richness	5	3.98	0.07	1.00	-620.56
Null	4	11.31	0.00	1.00	-625.26

Table A.16. AIC_C results from models describing the relationships between flower richness, urbanization, and bumble bee abundance sampled by surveys.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Flower Richness	5	0.00	0.42	0.42	-436.19
Flower Richness*Urbanization	7	0.74	0.29	0.71	-434.48
Flower Richness+Urbanization	6	0.78	0.29	1.00	-435.55
Null	4	61.45	0.00	1.00	-467.95

Table A.17. AIC_C results from models describing the relationships between flower richness, urbanization, and bumble bee richness sampled by surveys.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Flower Richness*Urbanization	7	0.00	0.37	0.37	-251.58
Flower Richness+Urbanization	6	0.24	0.33	0.69	-252.76
Flower Richness	5	0.34	0.31	1.00	-253.84
Null	4	40.06	0.00	1.00	-274.72

Table A.18. AIC_C results from models describing the relationships between flower richness, urbanization, and total bee abundance sampled by surveys.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Flower Richness	5	0.00	0.63	0.63	-640.42
Flower Richness+Urbanization	6	1.90	0.25	0.88	-640.33
Flower Richness*Urbanization	7	3.33	0.12	1.00	-640.00
Null	4	33.16	0.00	1.00	-658.03

Table A.19. AIC_C results from models describing the relationships between flower richness, urbanization, and total bee abundance sampled by bee bowls.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Flower Richness	5	0.00	0.26	0.26	-506.00
Flower Richness*Urbanization	7	0.08	0.25	0.52	-503.96
Null	4	0.15	0.25	0.76	-507.10
Flower Richness+Urbanization	6	0.22	0.24	1.00	-505.07

Table A.20. AIC_C results from models describing the relationships between flower richness, urbanization, and total bee richness sampled by bee bowls.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Null	4	0.00	0.29	0.29	-385.75
Flower Richness+Urbanization	6	0.17	0.27	0.56	-383.78
Flower Richness	5	0.18	0.27	0.83	-384.82
Flower Richness*Urbanization	7	1.08	0.17	1.00	-383.19

Table A.21. AIC$_C$ results from models describing the relationships between site type, urbanization, and honey bee abundance sampled by surveys.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Site Type+Urbanization	6	0.00	0.37	0.37	-618.77
Site Type*Urbanization	7	0.12	0.34	0.71	-617.79
Site Type+Urbanization^2	7	2.02	0.13	0.84	-618.74
Site Type*Urbanization^2	9	2.87	0.09	0.93	-617.06
Urbanization	5	5.01	0.03	0.96	-622.32
Site Type	5	5.34	0.03	0.98	-622.48
Urbanization^2	6	7.07	0.01	1.00	-622.31
Null	4	8.84	0.00	1.00	-625.26

Table A.22. AIC$_C$ results from models describing the relationships between site type, urbanization, and bumble bee abundance sampled by surveys.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Site Type+Urbanization^2	7	0.00	0.23	0.23	-450.77
Site Type+Urbanization	6	0.26	0.20	0.43	-451.94
Site Type	5	0.34	0.19	0.62	-453.02
Site Type*Urbanization^2	9	0.37	0.19	0.81	-448.85
Site Type*Urbanization	7	0.39	0.19	1.00	-450.96
Null	4	28.14	0.00	1.00	-467.95
Urbanization	5	29.49	0.00	1.00	-467.60
Urbanization^2	6	30.17	0.00	1.00	-466.90

Table A.23. AIC$_C$ results from models describing the relationships between site type, urbanization, and bumble bee richness sampled by surveys.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Site Type*Urbanization	7	0.00	0.37	0.37	-254.99
Site Type+Urbanization	6	1.17	0.21	0.58	-256.62
Site Type*Urbanization^2	9	1.30	0.19	0.77	-253.53
Site Type+Urbanization^2	7	1.94	0.14	0.91	-255.96
Site Type	5	2.89	0.09	1.00	-258.51
Null	4	33.25	0.00	1.00	-274.72
Urbanization	5	34.05	0.00	1.00	-274.09
Urbanization^2	6	35.35	0.00	1.00	-273.70

Table A.24. AIC$_C$ results from models describing the relationships between site type, urbanization, and total bee abundance sampled by surveys.

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Site Type	5	0.00	0.44	0.44	-644.48
Site Type+Urbanization	6	1.19	0.24	0.68	-644.03
Site Type+Urbanization^2	7	1.99	0.16	0.84	-643.39
Site Type*Urbanization	7	2.96	0.10	0.94	-643.88
Site Type*Urbanization^2	9	3.96	0.06	1.00	-642.27
Null	4	25.04	0.00	1.00	-658.03
Urbanization	5	26.89	0.00	1.00	-657.92
Urbanization^2	6	28.29	0.00	1.00	-657.59

Table A.25. AIC_C results from models describing the relationships between site type, urbanization, and total bee abundance sampled by bee bowls.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Site Type	5	0.00	0.31	0.31	-504.66
Site Type+Urbanization	6	0.32	0.26	0.57	-503.79
Site Type*Urbanization	7	1.58	0.14	0.71	-503.38
Site Type +Urbanization^2	7	2.33	0.10	0.81	-503.75
Null	4	2.82	0.08	0.89	-507.10
Urbanization	5	3.09	0.07	0.95	-506.21
Urbanization^2	6	5.12	0.02	0.98	-506.19
Site Type*Urbanization^2	9	5.16	0.02	1.00	-503.07

Table A.26. AIC_C results from models describing the relationships between site type, urbanization, and total bee richness sampled by bee bowls.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Site Type	5	0.00	0.31	0.31	-383.20
Site Type+Urbanization	6	0.15	0.29	0.60	-382.24
Site Type*Urbanization	7	1.80	0.13	0.73	-382.03
Site Type+Urbanization^2	7	2.22	0.10	0.83	-382.24
Null	4	3.05	0.07	0.90	-385.75
Urbanization	5	3.15	0.06	0.96	-384.78
Urbanization^2	6	5.21	0.02	0.98	-384.77
Site Type*Urbanization^2	9	5.95	0.02	1.00	-382.01

Table A.27. AIC_C results from models describing the relationships between site type, urbanization, and flower density.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Site Type	5	0.00	0.57	0.57	-1189.38
Site Type+Urbanization	6	1.99	0.21	0.79	-1189.34
Site Type+Urbanization^2	7	3.25	0.11	0.90	-1188.93
Site Type*Urbanization	7	3.84	0.08	0.98	-1189.22
Site Type*Urbanization^2	9	7.06	0.02	1.00	-1188.74
Null	4	20.82	0.00	1.00	-1200.82
Urbanization	5	22.26	0.00	1.00	-1200.51
Urbanization^2	6	23.66	0.00	1.00	-1200.18

Table A.28. AIC_C results from models describing the relationships between site type, urbanization, and flower richness.

Model	K	Delta AIC_C	AIC_C Weight	Cumulative Weight	LL
Site Type	5	0.00	0.58	0.58	578.43
Site Type+Urbanization	6	1.90	0.22	0.80	578.52
Site Type+Urbanization^2	7	3.40	0.11	0.91	578.80
Site Type*Urbanization	7	3.98	0.08	0.99	578.52
Site Type*Urbanization^2	9	7.60	0.01	1.00	578.80
Null	4	17.57	0.00	1.00	568.61
Urbanization	5	19.61	0.00	1.00	568.62
Urbanization^2	6	21.28	0.00	1.00	568.82

APPENDIX B: REFERENCES FOR BEE FUNCTIONAL TRAITS (CHAPTER 2)

References for bee functional traits in Table 2.1.

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APPENDIX C: AIC$_C$ TABLES (CHAPTER 3)

Table C.1. Results from first round of AIC$_C$ used to select covariates for models with beetle abundance as the dependent variable that use the full data set

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Temperature	4	0.00	0.99	0.99	-533.07
Wind	4	12.51	0.00	1.00	-539.34
Null	3	13.16	0.00	1.00	-540.68
Site Area	4	14.20	0.00	1.00	-540.17
Day	4	14.63	0.00	1.00	-540.39
Cloud Cover	4	15.21	0.00	1.00	-540.68

Table C.2. Results from first round of AIC$_C$ used to select covariates for models with fly abundance as the dependent variable that use the full data sets

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Temperature	4	0.00	0.49	0.49	-537.99
Day	4	1.32	0.25	0.74	-538.65
Null	3	2.93	0.11	0.86	-540.48
Site Area	4	4.42	0.05	0.91	-540.20
Wind	4	5.60	0.05	0.96	-540.29
Cloud Cover	4	4.96	0.04	1.00	-540.47

Table C.3. Results from first round of AIC$_C$ used to select covariates for models with true bug abundance as the dependent variable that use the full data set

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Null	3	0.00	0.30	0.30	-312.62
Day	4	0.89	0.20	0.50	-312.04
Cloud Cover	4	1.51	0.14	0.64	-312.35
Wind	4	1.71	0.13	0.77	-312.45
Temperature	4	1.95	0.11	0.89	-312.57
Site Area	4	1.97	0.11	1.00	-312.58

Table C.4. Results from first round of AIC$_C$ used to select covariates for models with wasp abundance as the dependent variable that use the full data set

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Day	4	0.00	0.48	0.48	-524.94
Null	3	2.07	0.17	0.64	-527.01
Site Area	4	2.73	0.12	0.77	-526.31
Cloud Cover	4	3.28	0.09	0.86	-526.58
Temperature	4	3.71	0.07	0.93	-526.80
Wind	4	3.91	0.07	1.00	-526.90

Table C.5. Results from first round of AIC$_C$ used to select covariates for models with lepidoptera abundance as the dependent variable that use the full data set

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Day	4	0.00	0.93	0.93	-447.93
Temperature	4	5.22	0.07	1.00	-450.54
Null	3	12.41	0.00	1.00	-455.17
Site Area	4	13.70	0.00	1.00	-454.78
Cloud Cover	4	14.11	0.00	1.00	-454.98
Wind	4	14.30	0.00	1.00	-455.08

Table C.6. Results from first round of AIC$_C$ used to select covariates for models with bee abundance as the dependent variable that use the full data set

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Temperature	4	0.00	0.86	0.86	-537.85
Cloud Cover	4	5.49	0.06	0.92	-540.59
Null	3	6.97	0.03	0.95	-542.36
Day	4	7.28	0.02	0.97	-541.49
Site Area	4	7.70	0.02	0.99	-541.70
Wind	4	8.55	0.01	1.00	-542.12

Table C.7. Results from first round of AIC$_C$ used to select covariates for models with total pollinator abundance as the dependent variable that use the full data set

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Temperature	4	0.00	0.99	0.99	-591.54
Cloud Cover	4	12.41	0.00	1.00	-597.75
Day	4	12.56	0.00	1.00	-597.83
Null	3	13.39	0.00	1.00	-599.27
Wind	4	15.24	0.00	1.00	-599.16
Site Area	4	15.43	0.00	1.00	-599.26

Table C.8. Results from first round of AIC$_C$ used to select covariates for models with fly abundance as the dependent variable that use the data set including only sites zoned by the HOLC

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Null	3	0.00	0.30	0.30	-79.76
Cloud Cover	4	1.09	0.17	0.48	-79.08
Temperature	4	1.72	0.13	0.60	-79.39
Wind	4	1.84	0.12	0.72	-79.46
Urbanization	4	2.25	0.10	0.82	-79.66
Day	4	2.41	0.09	0.91	-79.74
Site Area	4	2.44	0.09	1.00	-79.76

Table C.9. Results from first round of AIC$_C$ used to select covariates for models with lepidoptera abundance as the dependent variable that use the data set including only sites zoned by the HOLC

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Temperature	4	0.00	0.37	0.37	-52.90
Urbanization	4	0.46	0.30	0.66	-53.13
Null	3	2.39	0.11	0.78	-55.32
Wind	4	2.93	0.09	0.87	-54.37
Cloud Cover	4	3.81	0.06	0.92	-54.81
Day	4	4.41	0.04	0.97	-55.11
Site Area	4	4.74	0.03	1.00	-55.27

Table C.10. Results from first round of AIC$_C$ used to select covariates for models with bee abundance as the dependent variable that use the data set including only sites zoned by the HOLC

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Temperature	4	0.00	0.28	0.28	-84.38
Urbanization	4	0.53	0.22	0.50	-84.64
Null	3	0.72	0.20	0.70	-85.97
Cloud Cover	4	1.90	0.11	0.81	-85.34
Site Area	4	2.86	0.07	0.87	-85.81
Day	4	2.91	0.07	0.94	-85.84
Wind	4	3.13	0.06	1.00	-85.95

Table C.11. Results from first round of AIC$_C$ used to select covariates for models with lepidoptera abundance as the dependent variable that use the data set including only sites zoned by the HOLC

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Temperature	4	0.00	0.65	0.65	-90.17
Null	3	3.25	0.13	0.78	-93.02
Cloud Cover	4	5.20	0.05	0.83	-92.77
Urbanization	4	5.37	0.04	0.87	-92.86
Wind	4	5.39	0.04	0.91	-92.87
Site Area	4	5.52	0.04	0.96	-92.93
Day	4	5.54	0.04	1.00	-92.94

Table C.12. AIC$_C$ results from models describing relationships between race, income, urbanization, and beetle abundance

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Race	5	0.00	0.24	0.24	-528.58
Race+Urbanization	6	0.85	0.15	0.39	-527.96
Race*Urbanization	7	1.10	0.14	0.52	-527.03
Income+Race	6	1.54	0.11	0.63	-528.31
Urbanization	5	2.07	0.08	0.72	-529.62
Income*Urbanization+Race	8	2.34	0.07	0.79	-526.59
Income+Race+Urbanization	7	2.46	0.07	0.86	-527.71
Race*Urbanization+Income	8	2.99	0.05	0.91	-526.92
Income*Urbanization+Race*Urbanization	9	3.71	0.04	0.95	-526.21
Income+Urbanization	6	4.15	0.03	0.98	-529.61
Income*Urbanization	7	5.96	0.01	0.99	-529.46
Null	4	6.91	0.01	1.00	-533.07
Income	5	8.86	0.00	1.00	-533.01

Table C.13. AIC$_C$ results from models describing relationships between race, income, urbanization, and fly abundance

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Race+Urbanization	7	0.00	0.15	0.15	-534.86
Income*Urbanization	8	0.47	0.12	0.27	-534.03
Income+Race+Urbanization	8	0.98	0.09	0.37	-534.29
Null	5	1.06	0.09	0.46	-537.49
Income	6	1.07	0.09	0.55	-536.45
Race*Urbanization	8	1.38	0.08	0.63	-534.49
Income+Urbanization	7	1.39	0.08	0.70	-535.55
Race	6	1.66	0.07	0.77	-536.75
Income*Urbanization+Race	9	1.71	0.07	0.84	-533.58
Urbanization	6	2.06	0.05	0.89	-536.94
Race*Urbanization+Income	9	2.58	0.04	0.93	-534.02
Income+Race	7	2.62	0.04	0.97	-536.17
Income*Urbanization+Race*Urbanization	10	3.55	0.03	1.00	-533.42

Table C.14. AIC$_C$ results from models describing relationships between race, income, urbanization, and true bug abundance

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Race+Urbanization	5	0.00	0.23	0.23	-309.50
Race	4	0.09	0.22	0.46	-310.59
Race*Urbanization	6	1.88	0.09	0.55	-309.39
Income+Race+Urbanization	6	1.95	0.09	0.64	-309.43
Income+Race	5	1.98	0.09	0.73	-310.49
Null	3	2.10	0.08	0.81	-312.62
Income	4	2.92	0.05	0.86	-312.00
Race*Urbanization+Income	7	3.89	0.03	0.90	-309.35
Income*Urbanization+Race	7	3.96	0.03	0.93	-309.38
Urbanization	4	4.13	0.03	0.96	-312.60
Income+Urbanization	5	4.86	0.02	0.98	-311.93
Income*Urbanization+Race*Urbanization	8	5.86	0.01	0.99	-309.27
Income*Urbanization	6	6.54	0.01	1.00	-311.73

Table C.15. AIC$_C$ results from models describing relationships between race, income, urbanization, and wasp abundance

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Race*Urbanization	7	0.00	0.40	0.40	-519.87
Race*Urbanization+Income	8	2.10	0.14	0.55	-519.86
Race+Urbanization	6	2.68	0.11	0.65	-522.27
Income*Urbanization+Race*Urbanization	9	3.80	0.06	0.71	-519.64
Null	4	3.87	0.06	0.77	-524.94
Urbanization	5	4.29	0.05	0.82	-524.12
Income+Race+Urbanization	7	4.76	0.04	0.86	-522.25
Income*Urbanization	7	4.89	0.04	0.89	-522.31
Race	5	5.25	0.03	0.92	-524.60
Income*Urbanization+Race	8	5.56	0.03	0.94	-521.59
Income	5	5.73	0.02	0.97	-524.84
Income+Urbanization	6	5.82	0.02	0.99	-523.84
Income+Race	6	7.31	0.01	1.00	-524.58

Table C.16. AIC$_C$ results from models describing relationships between race, income, urbanization, and lepidoptera abundance

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Income+Race	7	0.00	0.21	0.21	-429.59
Null	5	0.73	0.15	0.36	-432.06
Race	6	0.91	0.13	0.49	-431.10
Income	6	1.76	0.09	0.58	-431.52
Income+Race+Urbanization	8	2.12	0.07	0.66	-429.59
Urbanization	6	2.32	0.07	0.72	-431.80
Race*Urbanization+Income	9	2.45	0.06	0.78	-428.68
Race*Urbanization	8	2.83	0.05	0.84	-429.94
Income+Urbanization	7	2.95	0.05	0.88	-431.06
Race+Urbanization	7	3.01	0.05	0.93	-431.10
Income*Urbanization+Race	9	4.26	0.03	0.96	-429.59
Income*Urbanization	8	4.53	0.02	0.98	-430.79
Income*Urbanization+Race*Urbanization	10	4.55	0.02	1.00	-428.65

Table C.17. AIC$_C$ results from models describing relationships between race, income, urbanization, and bee abundance

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Null	4	0.00	0.23	0.23	-537.85
Race	5	0.77	0.16	0.39	-537.20
Income+Race	6	1.54	0.11	0.49	-536.53
Income	5	1.66	0.10	0.59	-537.64
Urbanization	5	1.80	0.09	0.69	-537.71
Race+Urbanization	6	2.84	0.06	0.74	-537.19
Income*Urbanization	7	3.05	0.05	0.79	-536.23
Race*Urbanization	7	3.25	0.05	0.84	-536.33
Income+Urbanization	6	3.31	0.04	0.88	-537.42
Race*Urbanization+Income	8	3.51	0.04	0.92	-535.40
Income+Race+Urbanization	7	3.63	0.04	0.96	-536.52
Income*Urbanization+Race	8	4.15	0.02	0.98	-535.90
Income*Urbanization+Race*Urbanization	9	5.01	0.02	1.00	-535.08

Table C.18. AIC$_C$ results from models describing relationships between race, income, urbanization, and pollinator abundance

Model	K	Delta AIC$_C$	AIC$_C$ Weight	Cumulative Weight	LL
Income+Race	6	0.00	0.22	0.22	-588.32
Race	5	0.37	0.18	0.40	-589.55
Income+Race+Urbanization	7	1.21	0.12	0.51	-587.87
Race+Urbanization	6	1.59	0.10	0.61	-589.11
Income*Urbanization+Race	8	1.67	0.09	0.71	-587.04
Null	4	2.29	0.07	0.77	-591.54
Income*Urbanization	7	3.12	0.05	0.82	-588.82
Race*Urbanization+Income	8	3.32	0.04	0.86	-587.86
Race*Urbanization	7	3.57	0.04	0.90	-589.05
Income*Urbanization+Race*Urbanization	9	3.68	0.03	0.93	-586.97
Income	5	3.90	0.03	0.96	-591.31
Urbanization	5	4.29	0.03	0.99	-591.50
Income+Urbanization	6	5.81	0.01	1.00	-591.22

APPENDIX D: SEMI-STRUCTURED INTERVIEW GUIDE (CHAPTER 4)

1. How has the COVID-19 pandemic impacted your ability to spend time outdoors interacting with nature?

2. How has the pandemic impacted your ability to participate in citizen science projects?

3. Have you participated in any other citizen science projects during the pandemic in addition to the Urban Pollinators Project? If so, what projects and when did you participate?

4. How do you think that its online format impacted the accessibility of the Urban Pollinators Project during the pandemic? How did the online format increase the accessibility of participation? How did the online format decrease or limit the accessibility of participation?

5. Did you experience any challenges related to the use of Zoom and Google Docs during the trainings? If so, can you describe those challenges?

6. What could have been done to make this project easier to participate in?

7. Do you think that this project might have been more accessible to some groups of people than to others? If so, please explain.

8. Is there anything else that you would like to add?