INSECTS AND THE CITY:
SOCIO-ECOLOGICAL TRENDS IN URBAN AGROECOSYSTEMS

A thesis submitted in partial fulfillment
of the requirements for the degree of
Bachelor of Science with Honors
(Ecology, Evolution, and Biodiversity)
in the University of Michigan
2018

Thesis advisor: Professor John Vandermeer, EEB
Thesis readers: Professor Catherine Badgley, EEB
                     Professor Ivette Perfecto, SEAS

ACKNOWLEDGEMENTS

There are seemingly countless people without whom this written culmination of my undergraduate career would not have been possible. In an effort count them anyway, I'd like to thank the following groups and individuals for all they have contributed:

To my family: I cannot overstate how instrumental you have been in helping me become the person and scholar that I am today. Though I know you'd like me closer to home, I can always feel your love and support from afar.

To my mentors, colleagues, and friends in the Perfectomeer lab: I came to the University of Michigan without a clear idea of what I wanted to do with my academic career. When I applied to do research on a cool-sounding project about native bees, I could not have imagined how profoundly the experience would shape the trajectory of my life. John and Ivette, you never cease to be a source for my personal entertainment, professional aspirations, and intellectual curiosity. I am inspired by your abilities to synthesize rigorous scientific and political analyses, and by the family you have managed to create and maintain here at Michigan. I am so grateful I was lucky enough to be a part, and hope to emulate these qualities as I continue my academic journey. To my many lab mates through the past four years, thank you for helping foster such a welcoming environment and for your contributions to my education as a young agroecologist. Special thanks to fellow urban bee researchers including Gordon Fitch, Chatura Vaidya, and Austin Martin for your teaching and collaboration. Carolina Simao, even though you weren't my official UROP sponsor you acted as an invaluable mentor during my formative years in the lab; thank you for your expertise and your kindness. Finally, I would especially like to thank Paul Glaum. I cannot put into words how much your mentorship has meant to me. From the first day I entered your office to interview to be your UROP student to writing this thesis, your advice,

To all my friends: You have made me laugh, kept me sane, and helped me enjoy my time here at Michigan immensely. Special thanks to Emma Abed, Sara Lebow, and Max Lubell for always being down to share good food, hot takes, and wholesome memes. It was fun to have you along on the undergraduate thesis journey.

To my other comrades in Ann Arbor: members of NWAEG, Science for the People, UMSFP, Divest and Invest, and other community activists, thank you for helping me in the struggle to imagine and build a better future.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

INTRODUCTION

According to some scientists, the Earth has entered a new geological epoch dubbed the "Anthropocene" (Crutzen 2006). The defining characteristic of the Anthropocene is that human activity is now one of the primary drivers of global environmental change. Following this assessment, other researchers have sought to articulate to what extent human activity is threatening our ability to thrive, and which particular biophysical processes ought to be cause for concern (e.g. Rockström et al. 2009, Steffen et al. 2015). One such process that receives significant attention from ecologists, conservationists, and policymakers is biodiversity loss. With extinction greater than 1000 times background rates, species are disappearing at an alarming pace with profound consequences for ecosystem function including stability, nutrient capture, productivity, and efficiency (Cardinale et al. 2012). Relatedly, the anthropocene is marked by mass defaunation, with vertebrate abundance declining by 25% and invertebrates declining 45% on average since 1970 (Dirzo et al. 2014). Land use change at global and local scales is widely acknowledged as the primary driver of these losses; as natural ecosystems are converted to agriculture and urbanized to meet the needs of growing human populations, suitable habitat for various plant and animal species is destroyed (Foley et al. 2005, Grimm et al. 2008).

Agricultural expansion and intensification, particularly since the mid 20th century, is regularly criticized for its high environmental costs, including greenhouse gas emissions, distorted fluxes of limiting nutrients, consequential pesticide use, and habitat destruction (Tillman et al 2001). Some taxa of particular concern are insects and other arthropods, which provide valuable ecosystem services including pollination, pest control, and nutrient cycling that contribute to the sustainable functioning of both natural and managed ecosystems (Losey et al. 2006). Evidence suggests that insect communities are declining rapidly, even in natural areas.

For example, a recent study of German nature preserves found an estimated 76% decline in flying insect biomass since 1989 (Hallmann et al. 2017). The authors attribute this decline to habitat fragmentation and pesticide use associated with surrounding agricultural fields.

Concurrent with agricultural intensification, many landscapes have been urbanized as human communities concentrate and cities expand. Today more than 50% of the global population resides in urban areas, up from just 10% at the beginning of the 20th century (Grimm et al. 2008). Furthermore, urban land cover is predicted to nearly triple from year 2000 levels by 2030, with biodiversity hotspots projected to lose some of the most habitat area (Seto et al. 2012). Given these trends, it is increasingly important that ecologists turn their attention to cities. Yet the discipline of urban ecology is relatively young, and researchers are only starting to understand the role that urban environments play in the structure and function of biological communities.

Proposed conservation strategies for addressing habitat loss have historically—and perhaps erroneously—fallen into a debate over land sparing versus land sharing in the context of agricultural expansion. Proponents of land sparing argue that agriculture needs to continue to intensify in order to meet global food demand, producing more food on the same amount of land; correspondingly, large reserves of "pristine" natural habitat should be maintained to minimize the loss of biodiversity (Phalan et al. 2011, Egan et al. 2012). Critics of this dichotomous model counter that the land sparing framework fails to account for both the ecological interactions between farmed and unfarmed areas as well as on-the-ground realities of farmer livelihoods across the globe (Kremen 2015, Fisher et al. 2008, Perfecto & Vandermeer 2010). As others have pointed out (e.g. Lin & Fuller 2013, Goddard et al. 2010), a similar paradigm might equally apply to non-agricultural land uses with varying levels of intensity, such as urbanization.

In their 2009 book *Nature's Matrix*, authors Perfecto, Vandermeer, and Wright draw upon ecological theory to show that the quality of the landscape between natural habitat fragments is perhaps more important than the fragments themselves. Using a framework of metapopulations, they argue that the inevitability of local extinctions in "pristine" habitat patches means that the potential for migration between those patches (through the "matrix") is highly relevant for species persistence through time and space. This means that ignoring the type of agriculture occurring in a landscape, as proponents of land-sparing and intensification are wont to do, is hugely counterproductive for biodiversity conservation. They conclude that expanding agroecological farming practices by supporting grassroots, farmer-led social movements is the best way to shore up the twin goals of sufficient food production and biodiversity conservation (Perfecto et al. 2009).

While this account is convincing, it is heavily centered in the rural tropics, with little to say about either about the socio-ecological forces shaping agriculture in the Global North or the importance of urban areas embedded in "nature's matrix" and their relevance to biodiversity conservation. As mentioned above, urban development can result in spatial configurations remarkably similar to the landscape mosaics produced by agricultural expansion and intensification (Lin & Fuller 2013). If the agricultural matrix matters, so too should the urban one; it is thus important for ecologists, conservationists, and urban planners to understand how cities can be designed and managed to balance human needs with ecological realities. Indeed, previous work on green space in urban environments suggests that matrix quality may be valuable for conservation and ecosystem service provisioning. In a review of the relevance of urban vacant land to arthropods, Gardiner et al. (2013) found that vacant lots can be refuges for

rare species across many insect taxa in cities, but that more work is required to make reliable claims across various urban contexts.

While agriculture and urbanization are driving the problem of biodiversity loss, one part of the solution might ironically lie at their intersection. Urban agriculture, broadly defined as the practice of cultivating, processing, and distributing food in or around a village, town, or city, has been gaining both public and academic attention in recent decades (Palmer 2018). It is estimated that urban agriculture accounts for 15-20% of global food production (FAO), and it frequently comprises a large percentage of the green space in cities (Goddard et al. 2010). These factors could have strong positive implications for biotic communities, because urban agriculture tends to be intensive biologically rather than chemically or mechanically. Since plots are often small and crops are frequently for household or local consumption, agrobiodiversity in urban farms and gardens is typically higher than in industrial monocultures (Galuzzi 2010). The smaller scale also means that much of the management is conducted with human labor rather than fossil fuel-powered machines that both contribute to climate change (Pfeiffer 2006) and compact soil (Hamza & Anderson 2005). Additionally, the proximity of urban agriculture to high-density human systems disincentives the use of agrochemicals like pesticides and synthetic nitrogen fertilizers due to their deleterious public health consequences (Brown & Jameton 2000).

At its best, the multifunctional use of urban land for food production could incorporate all of these local practices while simultaneously reducing the imperative for agricultural expansion into rural areas (appeasing land-sparers) and increasing urban matrix quality (appeasing land-sharers). The aspects of this idealized system might be considered key features of "urban agroecology," an emerging field of study that synthesizes the science, practice, and politics of food production in cities. Although definitions are far from settled and vary widely with context,

urban agroecology can encompasses ecological inquiry and application as well as socio-political analysis of urban food landscapes in the pursuit of a more just and sustainable food system (RUAF 2017).

In this work, I seeks to elucidate some of the socio-ecological trends in small urban agroecosystems. I use gardens in southeast Michigan, USA as my study system, including sites in and around the city of Detroit. Detroit has a rich history of urban agriculture dating back to World War II Victory Gardens. In the context of post-war de-industrialization, there has been renewed interest using city land for food production as manifested in mayor Coleman Young's Farm-a-lot program in the 1970s, grassroots urban farming efforts in the early 2000s, and the passage of a city urban agriculture ordinance in 2013 (Pothukuchi 2015). Given that there is significant popular support (Colasanti et al. 2012) and physical capacity (Colasanti et al. 2016) for agriculture expansion in Detroit, it makes this an appropriate urban context for my research. Specifically, I focus on how physical and socio-economic factors affect community gardens and their insect ecology. In Chapter 1, I explore on how local- and landscape-level factors drive the abundance of particular taxonomic (flies, bees, beetles etc.) and functional groups (pollinators) in gardens along an urban-to-rural gradient. In Chapter 2, I utilize a subset of those gardens to provide a methodological case study relating ecological trends to the socio-economic demographics of garden neighborhoods to illustrate how interdisciplinary research can be used to address complex sustainability questions. I hope that these contributions can play a small part in helping to grow the field of urban agroecology, in service of a future that promotes the flourishing of both human and non-human life.

WORKS CITED

Brown, K. H., & Jameton, A. L. (2000). Public health implications of urban agriculture. *Journal of public health policy*, *21*(1), 20-39. doi:10.2307/3343472

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A. and Kinzig, A.P. (2012). Biodiversity loss and its impact on humanity. *Nature*, *486*(7401), 59. doi:10.1038/nature11148

Colasanti, K. J., Hamm, M. W., & Litjens, C. M. (2012). The city as an "agricultural powerhouse"? perspectives on expanding urban agriculture from Detroit, Michigan. *Urban geography*, *33*(3), 348-369. doi:10.2747/0272-3638.33.3.348

Colasanti, K. J., & Hamm, M. W. (2016). Assessing the local food supply capacity of Detroit, Michigan. *Journal of agriculture, food Systems, and community development*, 1(2), 41-58. doi:10.5304/jafscd.2010.012.002

Crutzen, P. J. (2006). The "anthropocene". In *Earth system science in the anthropocene* (pp. 13-18). Springer, Berlin, Heidelberg.

Dirzo, R., Young, H. S., Galetti, M., Ceballos, G., Isaac, N. J., & Collen, B. (2014). Defaunation in the Anthropocene. *Science*, *345*(6195), 401-406. doi:0.1126/science.1251817

Egan, J. F., & Mortensen, D. A. (2012). A comparison of land- sharing and land- sparing strategies for plant richness conservation in agricultural landscapes. *Ecological applications*, *22*(2), 459-471. doi:10.1890/11-0206.1

Fischer, J., Brosi, B., Daily, G.C., Ehrlich, P.R., Goldman, R., Goldstein, J., Lindenmayer, D.B., Manning, A.D., Mooney, H.A., Pejchar, L. and Ranganathan, J. (2008). Should agricultural policies encourage land sparing or wildlife- friendly farming?. *Frontiers in Ecology and the Environment*, *6*(7), 380-385. doi:10.1890/070019

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K. and Helkowski, J.H. (2005). Global consequences of land use. *science*, *309*(5734), 570-574. doi:10.1126/science.1111772

Galluzzi, G., Eyzaguirre, P., & Negri, V. (2010). Home gardens: neglected hotspots of agro-biodiversity and cultural diversity. Biodiversity and conservation, 19(13), 3635-3654. doi:10.1007/s10531-010-9919-5.

Gardiner, M. M., Prajzner, S. P., Burkman, C. E., Albro, S., & Grewal, P. S. (2014). Vacant land conversion to community gardens: influences on generalist arthropod predators and biocontrol services in urban greenspaces. *Urban ecosystems*, *17*(1), 101-122. doi:10.1007/s11252-013-0303-6

Goddard, M. A., Dougill, A. J., & Benton, T. G. (2010). Scaling up from gardens: biodiversity conservation in urban environments. *Trends in ecology & evolution*, *25*(2), 90-98. doi:doi.org/10.1016/j.tree.2009.07.016

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M. (2008). Global change and the ecology of cities. *science*, *319*(5864), 756-760. doi:10.1126/science.1150195

Hallmann, C. A., Sorg, M., Jongejans, E., Siepel, H., Hofland, N., Schwan, H., ... & Goulson, D. (2017). More than 75 percent decline over 27 years in total flying insect biomass in protected areas. *PloS one*, *12*(10), e0185809. doi.org/10.1371/journal.pone.0185809

Hamza, M. A., & Anderson, W. K. (2005). Soil compaction in cropping systems: A review of the nature, causes and possible solutions. *Soil and tillage research*, *82*(2), 121-145. doi.org/10.1016/j.still.2004.08.009

Kremen, C. (2015). Reframing the land- sparing/land- sharing debate for biodiversity conservation. *Annals of the New York Academy of Sciences*, *1355*(1), 52-76. doi:10.1111/nyas.12845.

Lin, B. B., & Fuller, R. A. (2013). Sharing or sparing? How should we grow the world's cities?. *Journal of Applied Ecology*, *50*(5), 1161-1168. doi:10.1111/1365-2664.12118

Losey, J. E., & Vaughan, M. (2006). The economic value of ecological services provided by insects. Bioscience, 56(4), 311-323. doi:10.1641/0006-3568(2006)56[311:TEVOES]2.0.CO;2

Palmer, L. (2018). Urban agriculture growth in US cities. *Nature Sustainability*, *1*(1), 5. doi:10.1038/s41893-017-0014-8

Perfecto, I., & Vandermeer, J. (2010). The agroecological matrix as alternative to the land-sparing/agriculture intensification model. Proceedings of the National Academy of Sciences, 107(13), 5786-5791. doi:10.1073/pnas.0905455107

Perfecto, I., Vandermeer, J., & Wright, A. (2009). *Nature's matrix: linking agriculture, conservation and food sovereignty*. Routledge.

Pfeiffer, D. (2009). *Eating fossil fuels: oil, food and the coming crisis in agriculture*. New Society Publishers.

Phalan, B., Onial, M., Balmford, A., & Green, R. E. (2011). Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science*, *333*(6047), 1289-1291. doi:10.1126/science.1208742

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F. S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J. and Nykvist, B. (2009). A safe operating space for humanity. *nature*, *461*(7263), 472. doi:10.1038/461472a

RUAF (2017). Urban Agroecology. *Urban Agriculture Magazine*, 33. http://www.ruaf.org/ua-magazine-no-33-urban-agroecology

Seto, K. C., Güneralp, B., & Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proceedings of the National Academy of Sciences*, *109*(40), 16083-16088. doi.org/10.1073/pnas.1211658109

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., ... & Folke, C. (2015). Planetary boundaries: Guiding human development on a changing planet. *Science*, *347*(6223), 1259855. doi:10.1126/science.1259855

Tilman, D., Fargione, J., Wolff, B., D'antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W.H., Simberloff, D. and Swackhamer, D. (2001). Forecasting agriculturally driven global environmental change. *Science*, *292*(5515), 281-284. 10.1126/science.1057544

CHAPTER 1

Insects differentially respond to changing conditions of urban community gardens across taxonomic groups and spatial scales

ABSTRACT

Urbanization is an important anthropogenic force with significant consequence for the abundance and diversity of biotic communities. Traits at both local- and landscape-scales play a role in determining the patterns of flora and fauna in cities. One taxon of particular concern is insects, which provide invaluable ecosystem services and indicate more general biodiversity trends, but are declining at an alarming rate. Since distinct landscape patterns can differ in their consequences for insects, it is important to study population trends across various geographic contexts. Here I study how the abundance of insects orders in urban gardens changes along an urban-to rural gradient around the "shrinking city" of Detroit, MI, evaluating the effects of impervious land cover and floral composition, as well as potential tradeoffs between taxonomically distinct pollinator groups. I find that insect abundance generally declines with increasing urban intensity and decreasing weed flower area, but that different scales of evaluation matter across insect orders, possibly attributable to varied life history traits. Furthermore, there is no evidence of pollinator tradeoffs across garden sites. These results suggest new research directions and conservation strategies in urban agroecosystems.

INTRODUCTION

      Urbanization is a major human-driven land use change that mediates the abundance and

diversity of organisms within landscapes. In particular, urbanization tends to cause biotic

homogenization because of changing local and landscape traits (McKinney 2006). The relative

importance of these two scales of environmental drivers, from the variety of vegetation or soil

type at a particular site to forest proximity or amount of green space in the surrounding area, can

vary depending on the particular taxa in question. Local and landscape drivers of biotic

abundance and diversity in urban environments have been demonstrated for birds (Melles et al

2003, Clergeau et al. 1998), small mammals (Baker et al. 2003) amphibians (Hamer & Paris

2011), and various insect groups (Egerer et al. 2017, Philpott et al. 2014, Matteson 2007).

      Studying the dynamics of insect communities in urban landscapes is of particular interest

because of their vital role in both top-down and bottom-up ecosystem processes (Yang & Gratton

2014, Weisser & Siemann 2008). Studies have demonstrated alarming declines in insect biomass

over time (Hallmann et al. 2017), and in the abundance of particular taxonomic groups including

bees (Cameron et al. 2011, Potts et al. 2010, National Resource Council 2007), butterflies (Habel

et al. 2016), and moths (Conrad et al. 2006). Understanding the drivers of these losses is critical

to informing efforts to conserve insects and ultimately preserve ecosystem function.

Furthermore, arthropods' diverse life history strategies and relatively fast response to

environmental change make them a useful indicator for more general trends in urban biotic

communities (McIntyre 2000).

      The growing attention paid to urban agriculture in recent decades, particularly in the

United States, makes gardens an increasingly salient socio-ecological unit for evaluating the

dynamics of insect populations in cities (Lin et al. 2015). Gardens have the potential to serve as

resource hotspots that help insects populations persist through space and time despite an otherwise inhospitable urban matrix; in turn, insects are directly relevant to urban agriculture outcomes due to their potential roles as pest, pollinator, and/or biological control agent. This is to say that those concerned with insect conservation should pay attention to urban gardens, and those seeking to create sustainable urban food systems must consider insects.

To date, much of the work on insect abundance and diversity in urban gardens has focused on bees due to the important pollination services they provide (e.g. Matteson 2008, Fortel et al. 2014, Pardee & Philpott 2014, Glaum et al. 2017). Yet the bulk of urban insect communities are non-bee species, which in addition to pollination can provide other relevant ecosystem services and disservices. Understanding how urbanization affects the abundance of insects generally, as well as particular non-bee insect groups, is thus relevant for both conservation and agriculture. Research on broader insect communities in urban gardens has found varying local and landscape patterns depending on geographic context. In Phoenix, AZ McIntyre et al. (2001) found local land use traits drove changes in ground-dwelling arthropod community composition but not richness or abundance; in the California central coast, five arthropod groups responded to urbanization differently across spatial scales (Egerer 2017); in Toledo, OH flying insect and spider abundance have been negatively correlated with landscape-level measures of urban intensity (Lagucki et al. 2017) while ground-dwelling arthropods were overwhelmingly affected by local factors (Philpott et al. 2014). The diversity of outcomes in these examples demonstrates the necessity of evaluating insect community dynamics in more cities before general trends can be discerned. In particular, "shrinking cities" might provide a unique landscape context for urban arthropods because of abundant vacant land that can serve as a key resource (Gardiner et al. 2013). Indeed, Glaum et al. (2017) found that the southeast

Michigan bumble bee community declined in abundance and diversity along an urbanization gradient outside of Detroit, but that sites in the city supported robust bumble bee communities, potentially attributable to the high amount of vacant land there. Accordingly, it is worthwhile to consider how such landscape patterns might affect other insect groups.

At the same time, the varying scales at which insects operate for foraging, dispersal, etc. mean that local traits, such as the flowering plant community, could be much more important for certain taxa. Furthermore, even when two groups of organisms are functionally similar (i.e. provide the same ecosystem service), particular conditions could impact them differently because of idiosyncratic life history traits. For example, distinct larval feeding requirements for bees versus hoverflies—family *Syrphidae*, another important but underappreciated group of pollinators and biocontrol agents (Ssymank et al. 2008)—could mean that sites supporting one do not successfully support the other. In such cases we might see trade-offs among groups throughout the landscape. Such a trade-off between wild bees and hoverflies has been demonstrated in an agricultural matrix (Jauker et al. 2009) but has yet to be tested directly in urban environments.

To elucidate these dynamics, this study analyzes local and landscape drivers for the broader insect community captured in the urban bee studies of Glaum et al. 2017 and Fitch et al. (in review). Specifically, it addresses three questions:

1. How does increasing urban intensity at the landscape scale affect community garden insect abundance across taxonomic groups?

2. How does local floral resource abundance and composition in community gardens affect insect abundance across taxonomic groups?

3. Is there evidence of trade-offs between taxonomically distinct insect pollinators in urban gardens?

METHODS

Study system

18 community food gardens and 1 natural reserve area in southeast Michigan spanning Livingston, Washtenaw, and Wayne counties along an urban-rural gradient were selected for sampling (Figure 1.1). Urban gradient analysis is a commonly used and effective technique for studying the impacts of urbanization on biotic communities (McDonnell & Hahs 2008). Not only do urban-rural gradient studies provide insight into the spatial dynamics of insects in and around cities, they also can serve as a time proxy for the effects of anticipated urban expansion (Bates et al. 2011). Gardens were selected with the help of two garden management organizations, Project Grow in Ann Arbor, MI and Growing Hope in Ypsilanti, MI. All gardens observed organic growing practices prohibiting the use of synthetic fertilizers and pesticides.

GIS and urban intensity analysis

To quantify the intensity of urban development around sample sites, geographic information systems (GIS) and data from the 2011 National Land Cover Database (NLCD) were used to create profiles of the proportion of impervious surface in areas around samples sites. Impervious surface refers to roads, buildings, and other structures that effectively blanket natural land cover, making it impenetrable. Following the suggestion of McKinney 2008 to define urban landscapes as those with greater than 50% impervious surface, NLCD "medium" (50-79%) and "high" (80-100%) classifications were combined to obtain measures of urbanity at 500 m, 1 km,

1.5 km, and 2 km radii from each sampling site. These areas were then divided by the total area in the buffer to obtain proportions used in analysis.

Field sampling and identification

Insects were sampled over the course of ten weeks at the beginning of the 2014 growing season. Sampling was conducted once in May, twice in June, and twice in July, for a total of five samplings with the exception of one site where sampling did not begin until June. Sampling days were selected to minimize confounding effects of inclement weather. Insects were caught using pan traps and bottle traps. At each sampling site, nine 2 oz. (59 mL) plastic bowls coated in blue, yellow, and white UV-reflective paint (three of each color) were arranged in a rectangular grid. Pans were filled with a solution of water and scentless soap to attract and capture flower visiting insects (Campbell et al. 2007). At sites covered with high vegetation, pan traps were mounted on PVC pipes to make them visible to flying insects. Bottle traps were constructed from plastic 2-liter bottles and filled with a solution of wine, vinegar, and sugar to attract insects feeding on fermenting organic material. For each sampling date, traps were placed at each field site before 10:00 AM and collected 24 hours later. Upon collection, insect samples were sorted and stored in 70% ethanol solution for later identification. Bees were identified to species using the keys at DiscoverLife.org (Ascher & Pickering 2015), with additional identification completed by experts Dr. Jason Gibbs (University of Manitoba, Winnipeg, Canada) and Jamie Pawelek (Wild Bee Garden Design, formerly University of California Berkeley, USA). Other insects were identified to the order or family level following the guides in Tripplehorn & Johnson (2005).

Over the course of the growing season, data on the flowering plant community within a 20 m radius of the center of the trap installation at each site were collected. Flower species were

identified in the field or photographed and described for later identification, and counts of each species were estimated using a logarithmic scale. Counts were then used in tandem with species average measurements of floral geometry to calculate the area of open inflorescence ("floral area"). To account for how different types of flowers might differently affect insect communities, flower species were further grouped into "native," "introduced," and "invasive" as well as the categories "crop," "weed," and "ornamental" based on literature review, and floral areas were calculated for each category.

Statistical analyses

All data analysis was conducted using RStudio version 1.0.153. To account for having one fewer sampling date at one site, sampling dates were treated as replicates for each site and averaged to produce the mean data points used in analyses. In addressing all three study questions, linear regressions were used to correlate relevant predictor and response variables. To investigate how increasing urban intensity affected insect abundance (question 1), I treated the proportion of impervious surface at each buffer radius as independent variables and mean insect abundance across various taxonomic groups as dependent variables. To investigate the effect of local flower abundance and composition (question 2), insect data were again treated as dependent while the aggregate and relative abundance of separate floral types were used as predictors. Finally, to look for potential tradeoffs among pollinators (question 3) the ratio of syrphid flies to bees were regressed against urban intensity measurements, and abundances of those groups were regressed against each other.

Dependent variables were tested for normality using the Shapiro-Wilks test. Skewed variables with non-normal distributions were square root transformed, as is conventional for

count data (McDonald 2009). All transformed variables were reexamined using normal Q-Q plots and the Shapiro-Wilks test to verify better fits to normality. In assessing the statistical significance of linear models, α was set to 0.05, and Akaike information criterion (AIC) values were used to compare model fits across spatial scales (as in Egerer et al. 2017, Bates et al. 2011).

RESULTS

Overall I identified 6,243 specimens from 9 insect orders during the sampling period. Of these, the vast majority (4,187 individuals, 67.1%) were flies (order Diptera), followed by bees, wasps, and ants (order Hymenoptera, 1257 individuals, 20.1%), beetles (order Coleoptera, 502 individuals, 8.0%), true bugs (order Hemiptera, 161 individuals 2.5%), and butterflies and moths (order Lepidoptera, 72 individuals 1.2%). Grasshoppers and crickets (order Orthoptera, 43 individuals), thrips (order Thysanoptera, 15 individuals), earwigs (order Dermaptera, 3 individuals), and caddisflies (order Trichoptera, 3 individuals) all made up less than 1% of sampled insect specimens (Figure 1.2).

Urban intensity (landscape-scale)

Across the urban gradient, mean insect abundance was significantly negatively correlated with increasing proportion of impervious surface around sampling sites for all buffer radii, with the best model at the 500 m buffer (Figure 1.3, Table 1.1).

To investigate how urban intensity might differentially affect insects across taxonomic groups, insect data was separated and analyzed by order across the four scales of urban intensity measurements. Significant negative relationships were found at all four scales for insects in aggregate, Hymenoptera, Coleoptera, and Hemiptera, but at no scale for Diptera alone. Model

fits decreased with increasing buffer radius for insects in aggregate and Coleoptera, but increased for Hymenoptera and Hemiptera (Table 1). Other insect orders were not sufficiently represented in samples to analyze independently, and were thus not included in separate linear models.

Floral area (local-scale)

Mean floral area was not found to be a significant predictor of mean insect abundance. When broken down by insect order, only Diptera abundance was significantly correlated with floral area ($p = 0.0136$, $R^2 = 0.2676$); as mean floral area decreased, so did mean Diptera abundance.

In considering the relationships between flower categories and insect variables, the mean area of "weed" flowers showed a significant positive correlation with mean insect abundance ($p = 0.0051$, $R^2 = 0.3417$). This relationship was driven by Diptera ($p = 0.0372$), Coleoptera ($p = 0.0220$), and Hemiptera ($p = 0.0237$) species, but did not hold for Hymenoptera ($p = 0.3672$). No other floral area category (crop, ornamental, native, introduced, or invasive) was a significant predictor of mean insect abundance in aggregate or when separated by order.

Pollinator tradeoffs

Nine insect families identified in my samples are well-documented contributors to pollination services. This includes five families of bees (Andrenidae, Apidae, Halictidae, Colletidae, and Megachilidae), one family of wasps (Vespidae), and two families of flies (Syrphidae and Bombyliidae). Of these only bee families and hoverflies flies (Syrphidae) were abundant enough to consider separately in analyses (735 bees, 217 hoverflies). Bee were

dominated by the family was Halictidae, with 567 individuals representing 77.1% of all bee samples.

Bees did not respond significantly to any land cover or floral variables as a group or when separated by families, while hoverflies responded negatively to increasing urban development (significant only at the 500 m buffer, p = 0.02741, R^2 = 0.2113) but no local floral variables. Despite their differing individual responses, the ratio of hoverfly to bee pollinators did not significantly change with increasing urban intensity. In fact, bee abundance was positively correlated with hoverfly abundance across sampling sites (p = 0.0016, R^2 = 0.4209) (Figure 4).

DISCUSSION

Overall increasing urban intensity was found to negatively impact insect abundance across taxonomic groups, although the extent of this impact varied for particular orders and depended on the spatial scale at which urban intensity was measured. This is consistent with the findings of a similar study conducted in the greenspaces of Toledo, OH, which documented declines in the aggregate flying Arthropod community with increasing impervious surface (Lagucki et al. 2017). Interestingly, Lagucki et al. found significant (α = 0.05) order-specific responses to impervious surface measurements only for Diptera and Araneae (spiders, not considered here), whereas the present study found significant responses for all sufficiently abundant orders *except* Diptera (i.e. Hymenoptera, Coleoptera, and Hemiptera). However, Lagucki et al. did find significant order-specific responses for Lepidoptera and Hymenoptera when using distance from the city center, another metric of urban intensity, as the predictor variable. Taken together these results strongly demonstrate that landscape conditions can play an

important role for flying insect populations in and around cities, even in resource-rich green spaces.

Although the observed decline in aggregate insect abundance with increasing urbanization is intuitive given the expected relationship between human development and habitat/resource availability, it does stand in contrast to some emerging ideas about the unique ecological context of shrinking cities and vacant land (Gardiner et al. 2013, Haase 2008). While Glaum et al. (2017) found a high number of diverse bumble bees in the most developed urban environments using the same study system, I failed to see such trends for other insect taxa. While these findings do not invalidate the hypothesis that vacant land can be a valuable resource to insects in cities, it does challenge the idea that cities with abundant vacant land represent a qualitatively different landscape paradigm for insect populations. At the same time, this study did not explicitly address important landscape variables like connectivity and patch size (Goddard et al. 2010). More mechanistic research is needed to elucidate how and why different scales of landscape complexity result in unique outcomes for particular insect groups.

Similar to Egerer et al. (2017), I found that different insect groups responded most strongly to urban development measured at different spatial scales, and I hypothesize that this has to do with the varying mobility of groups throughout their life cycles (Keitt et al. 1997). Although Diptera did not significantly respond to impervious land cover at any spatial scale measured, linear models approached significance and AIC value decreased with decreasing buffer radius. This could be because local factors tend to be more important for fly populations (Fuller et al. 2017), though few studies have comprehensively compared the relative importance of local and landscape factors for flies. Land cover measures at a finer, more localized scale, as well as disaggregating fly abundance data by species or functional group, could possibly reveal

clearer trends in fly abundance. Indeed, hoverflies—many of which are known to have foraging and dispersal ranges less than a few hundred meters (Lövei et al. 2009, Wratten et al. 2003)—significantly decreased with urban intensity measured at the smallest buffer radius. Other Diptera groups are likely respond differentially according to body size, lifespan, and other life history traits.

Nearly 75% of my Hymenoptera sample was comprised of bees and wasps. These tend to be highly mobile organisms that have wide ranges for foraging and dispersal, especially large-bodied species (Greenleaf et al. 2007). Many of these species are pollinators, and landscape context can be particularly important for these populations because of their patterns of resource use (Kremen et al. 2007). This expectation is consistent with the finding that urban intensity is a significant predictor of hymenoptera abundance when measured at both small and large scales within the expected foraging range of hymenopteran pollinators, generally up to 2 km. At measures of urban intensity greater than 2 km, we might expect model fits to decline.

The species in the remaining orders, Coleoptera and Hemiptera, are incredibly diverse in their morphology, behavior, and life history traits, making generalizations about either group difficult. My results show that urban intensity as measured at the 500 m buffer was the best predictor of Coleoptera abundance, but no clear trend in model fit is evident with increasing buffer radius. In contrast, urban intensity as measured at the 2 km radius was the best predictor of Hemiptera abundance, and model fit declined with decreasing scale. Literature on hemipteran land use patterns is sparse compared to other groups and tends to be highly variable, but some of the best studied species such as aphids and leafhoppers are known to disperse over long distances using wind (Johnson 1969). In general, ecologists and entomologists should seek to understand the particular life strategies and dispersal patterns for taxa of interest in order to determine what

spatial scales are most appropriate for discerning the effect of landscape factors on insect populations.

Surprisingly few floral variables had a measurable impact on insect abundance, even for taxa such as bees and hoverflies that depend on floral resources like nectar and pollen. This could be due in part to the scale of the floral measurements taken. Although most all pollinators have foraging ranges of at least a few hundred meters (Greenleaf et al. 2007), floral area was measured only within a 20 m radius of the trap setup. The only group for which overall floral area had a measurable impact on abundance was Diptera. Although the association between flies and flowers is often overlooked, many Diptera species do utilize floral resources (Larson et al. 2001). Furthermore, even species that may not depend on flowers directly could be responding to changes in vegetation density, for which floral area measurements are a proxy. The finding that local vegetation was important for flies is also consistent with my expectations from landscape-scale analyses above.

The only other measured floral variable found to significantly correlate with insect abundance was the area of "weed" flowers. Although "weed" is a socially defined category applied to any plant deemed undesirable to human managers, weeds can have profound ecological relevance because they contribute to the biological diversity of agroecosystems, especially for insects (Marshall et al. 2003, Nicholls & Altieri 2013, Andow 1991, Shelton 1983). However, it is important to note that in this study system weed floral area was significantly correlated with landscape-scale measurements of urban intensity. Anecdotally, the urban gardens sampled in denser urban environments tended to employ raised beds more frequently than those in more rural areas, resulting in less non-crop vegetation present in the areas where insect and floral sampling occurred. Although my study design does not allow me to

disentangle the effects of weeds and impervious surface, this observation highlights the ways that local and landscape factors are not perfectly independent. In this case, it is possible that decreasing weed prevalence is one mechanism by which urbanization affects insect abundance in cities. Future work on urban biodiversity should pay attention to the ways that that the local- and landscape-scales can be co-constitutive, rather than treating them dichotomously.

Our results do not show any evidence of tradeoffs between bee and hoverfly pollinators in the urban landscape context, even though hoverflies showed a significant linear decline with urban intensity while bees did not. The only other study of changing bee and hoverfly assemblages along an urban-rural gradient found that bee abundance responded to land cover but hoverflies only responded to flower abundance (Bates et al. 2011). Although the study did not look explicitly, their results also do not seem to find a tradeoff between bees and hoverflies. These findings in the urban context stand in contrast to previous work in agricultural landscapes that showed bee abundance in agricultural fields decreased with distance from natural ("source") habitat while hoverfly abundance increased; this is attributed to the differing local resource needs of bees and hoverflies for breeding (Jauker et al. 2009). In my study system, there was a significant correlation of bee abundance with hoverfly abundance across sample sites. This suggest that conditions supporting abundant bee populations in urban gardens are also conditions that support abundant hoverfly populations. I hypothesize the conditions in question are at the local scale, illustrated by the fact that the site supporting most abundant bee community by far also supported the most abundant syrphid fly community (rightmost data point in Figure 1.4), even though it was at a moderate level of urban intensity and did not support the most abundant insect community generally. While these hypothetical conditions went unmeasured in the present study, this result suggests that urban gardens can be managed to enhance the abundance of varied

pollinator taxa. Future urban garden research should test management practices mechanistically in order to operationalize ecological knowledge for pollinator conservation in cities.

Our findings demonstrate the significant role of urbanization in shaping biotic communities. Conditions at landscape and local scales, as well as the interactions between them, can have important consequences for insects in cities, but for the purpose of conservation it is important to remember that such conditions do not affect all insects equivalently. In some cases there will be opportunity costs to particular landscape designs and garden management practices, so understanding how these decisions impact relevant insect groups will be vital. Although I did not have the resources or expertise to identify all insect specimens to ideal levels of taxonomic specificity (i.e. functional group, feeding guild, species), the trends presented here can provide a valuable starting point for future research into the nuanced interactions between humans, insects, and the built environment.

WORKS CITED

Andow, D. A. (1991). Vegetational diversity and arthropod population response. *Annual review of entomology*, *36*(1), 561-586. doi:10.1146/annurev.en.36.010191.003021

Ascher, J. S., & Pickering, J. (2015). Discover Life bee species guide and world checklist (Hymenoptera: Apoidea: Anthophila). http://www.discoverlife.org/mp/20q?guide=Apoidea_species.

Baker, P. J., Ansell, R. J., Dodds, P. A., Webber, C. E., & Harris, S. (2003). Factors affecting the distribution of small mammals in an urban area. *Mammal review*, *33*(1), 95-100. doi:10.1046/j.1365-2907.2003.00003.x

Bates, A. J., Sadler, J. P., Fairbrass, A. J., Falk, S. J., Hale, J. D., & Matthews, T. J. (2011). Changing bee and hoverfly pollinator assemblages along an urban-rural gradient. *PloS one*, *6*(8), e23459. doi:10.1371/journal.pone.0023459

Cameron, S. A., Lozier, J. D., Strange, J. P., Koch, J. B., Cordes, N., Solter, L. F., & Griswold, T. L. (2011). Patterns of widespread decline in North American bumble bees. *Proceedings of the National Academy of Sciences*, *108*(2), 662-667. doi:10.1073/pnas.1014743108

Campbell, J. W., & Hanula, J. L. (2007). Efficiency of Malaise traps and colored pan traps for collecting flower visiting insects from three forested ecosystems. *Journal of Insect Conservation*, *11*(4), 399-408.

Clergeau, P., Savard, J. P. L., Mennechez, G., & Falardeau, G. (1998). Bird abundance and diversity along an urban-rural gradient: a comparative study between two cities on different continents. *Condor*, 413-425. doi:10.2307/1369707

Conrad, K. F., Warren, M. S., Fox, R., Parsons, M. S., & Woiwod, I. P. (2006). Rapid declines of common, widespread British moths provide evidence of an insect biodiversity crisis. *Biological conservation*, *132*(3), 279-291. doi:10.1016/j.biocon.2006.04.020

Egerer, M. H., Arel, C., Otoshi, M. D., Quistberg, R. D., Bichier, P., & Philpott, S. M. (2017). Urban arthropods respond variably to changes in landscape context and spatial scale. *Journal of urban ecology*, *3*(1), jux001. doi: 10.1093/jue/jux001

Fortel, L., Henry, M., Guilbaud, L., Guirao, A. L., Fortel, L., Henry, M., Guilbaud, L., Guirao, A.L., Kuhlmann, M., Mouret, H., Rollin, O. and Vaissière, B.E. (2014). Decreasing abundance, increasing diversity and changing structure of the wild bee community (Hymenoptera: Anthophila) along an urbanization gradient. *PloS one*, *9*(8), e104679. doi:10.1371/journal.pone.0104679.

Fuller, L., Fuentes- Montemayor, E., Watts, K., Macgregor, N. A., Bitenc, K., & Park, K. J. (2017). Local- scale attributes determine the suitability of woodland creation sites for Diptera. *Journal of applied ecology*. doi:10.1111/1365-2664.13035

Gardiner, M. M., Burkman, C. E., & Prajzner, S. P. (2013). The value of urban vacant land to support arthropod biodiversity and ecosystem services. *Environmental entomology*, 42(6), 1123-1136. doi:10.1603/EN12275

Glaum, P., Simao, M. C., Vaidya, C., Fitch, G., & Iulinao, B. (2017). Big city Bombus: using natural history and land-use history to find significant environmental drivers in bumble-bee declines in urban development. *Royal Society open science*, *4*(5), 170156. doi:10.1098/rsos.170156

Goddard, M. A., Dougill, A. J., & Benton, T. G. (2010). Scaling up from gardens: biodiversity conservation in urban environments. *Trends in ecology & evolution*, *25*(2), 90-98. doi:doi.org/10.1016/j.tree.2009.07.016

Greenleaf, S. S., Williams, N. M., Winfree, R., & Kremen, C. (2007). Bee foraging ranges and their relationship to body size. *Oecologia*, *153*(3), 589-596. doi:10.1007/s00442-007-0752-9

Haase, D. (2008). Urban ecology of shrinking cities: an unrecognized opportunity?. *Nature and culture*, *3*(1), 1. doi:10.3167/nc.2008.030101.

Habel, J. C., Segerer, A., Ulrich, W., Torchyk, O., Weisser, W. W., & Schmitt, T. (2016). Butterfly community shifts over two centuries. *Conservation biology*, *30*(4), 754-762. doi:10.1111/cobi.12656

Hallmann, C.A., Sorg, M., Jongejans, E., Siepel, H., Hofland, N., Schwan, H., Stenmans, W., Müller, A., Sumser, H., Hörren, T. and Goulson, D. (2017). More than 75 percent decline over 27 years in total flying insect biomass in protected areas. *PloS one*, *12*(10), e0185809. doi:10.1371/journal.pone.0185809

Hamer, A. J., & Parris, K. M. (2011). Local and landscape determinants of amphibian communities in urban ponds. *Ecological applications*, *21*(2), 378-390. doi:10.1890/10-0390.1

Jauker, F., Diekötter, T., Schwarzbach, F., & Wolters, V. (2009). Pollinator dispersal in an agricultural matrix: opposing responses of wild bees and hoverflies to landscape structure and distance from main habitat. Landscape Ecology, 24(4), 547-555. doi:10.1007/s10980-009-9331-2.

Johnson, C. G. (1969). *Migration and dispersal of insects by flight*. Methuen And Co. Ltd.; London.

Keitt, T., Urban, D., & Milne, B. (1997). Detecting critical scales in fragmented landscapes. *Conservation ecology*, *1*(1). http://www.consecol.org/vol1/iss1/art4/

Kremen, C., Williams, N. M., Aizen, M. A., Gemmill- Herren, B., LeBuhn, G., Minckley, R., ... & Vázquez, D. P. (2007). Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land- use change. *Ecology letters*, *10*(4), 299-314. doi:10.1111/j.1461-0248.2007.01018.x

Lagucki, E., Burdine, J. D., & McCluney, K. E. (2017). Urbanization alters communities of flying arthropods in parks and gardens of a medium-sized city. *PeerJ*, *5*, e3620. doi:10.7717/peerj.3620

Lin, B. B., Philpott, S. M., & Jha, S. (2015). The future of urban agriculture and biodiversity-ecosystem services: challenges and next steps. *Basic and applied ecology*, *16*(3), 189-201. doi:10.1016/j.baae.2015.01.005

Marshall, E. J. P., Brown, V. K., Boatman, N. D., Lutman, P. J. W., Squire, G. R., & Ward, L. K. (2003). The role of weeds in supporting biological diversity within crop fields. *Weed research*, *43*(2), 77-89. doi:10.1046/j.1365-3180.2003.00326.x

Matteson, K.C. (2007). Diversity and conservation of insects in urban gardens: theoretical and applied implications. *ETD Collection for Fordham University*. AAI3286423. https://fordham.bepress.com/dissertations/AAI3286423

Matteson, K. C., Ascher, J. S., & Langellotto, G. A. (2008). Bee richness and abundance in New York City urban gardens. *Annals of the Entomological Society of America*, *101*(1), 140-150. doi:10.1603/0013-8746(2008)101[140:BRAAIN]2.0.CO;2.

McDonald, J. H. (2009). Data transformations. *Handbook of biological statistics* (Vol. 2, pp. 160-164). Baltimore, MD: Sparky House Publishing.

McDonnell, M. J., & Hahs, A. K. (2008). The use of gradient analysis studies in advancing our understanding of the ecology of urbanizing landscapes: current status and future directions. Landscape Ecology, 23(10), 1143-1155. doi:10.1007/s10980-008-9253-4

McIntyre, N. E. (2000). Ecology of urban arthropods: a review and a call to action. *Annals of the Entomological Society of America*, *93*(4), 825-835. doi:10.1603/0013-8746(2000)093[0825:EOUAAR]2.0.CO;2

McIntyre, N. E., Rango, J., Fagan, W. F., & Faeth, S. H. (2001). Ground arthropod community structure in a heterogeneous urban environment. *Landscape and urban planning*, *52*(4), 257-274. doi:10.1016/S0169-2046(00)00122-5

McKinney, M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biological conservation*, *127*(3), 247-260. doi:10.1016/j.biocon.2005.09.005

McKinney, M. L. (2008). Effects of urbanization on species richness: a review of plants and animals. *Urban ecosystems*, *11*(2), 161-176. doi:10.1007/s11252-007-0045-4

Melles, S., Glenn, S., & Martin, K. (2003). Urban bird diversity and landscape complexity: species–environment associations along a multiscale habitat gradient. *Conservation ecology*, *7*(1). http://www.consecol.org/vol7/iss1/art5/

National Research Council. (2007). *Status of pollinators in North America*. National Academies Press.

Nicholls, C. I., & Altieri, M. A. (2013). Plant biodiversity enhances bees and other insect pollinators in agroecosystems. A review. *Agronomy for sustainable development*, *33*(2), 257-274. doi:10.1007/s13593-012-0092-y

Pardee, G. L., & Philpott, S. M. (2014). Native plants are the bee's knees: local and landscape predictors of bee richness and abundance in backyard gardens. *Urban ecosystems*, *17*(3), 641-659. doi:10.1007/s11252-014-0349-0.

Philpott, S. M., Cotton, J., Bichier, P., Friedrich, R. L., Moorhead, L. C., Uno, S., & Valdez, M. (2014). Local and landscape drivers of arthropod abundance, richness, and trophic composition in urban habitats. *Urban Ecosystems*, *17*(2), 513-532. doi:10.1007/s11252-013-0333-0.

Potts, S. G., Roberts, S. P., Dean, R., Marris, G., Brown, M. A., Jones, R., ... & Settele, J. (2010). Declines of managed honey bees and beekeepers in Europe. *Journal of apicultural research*, *49*(1), 15-22. doi:10.3896/IBRA.1.49.1.02

Shelton, M. D., & Edwards, C. R. (1983). Effects of weeds on the diversity and abundance of insects in soybeans. *Environmental Entomology*, *12*(2), 296-298. doi:10.1093/ee/12.2.296

Ssymank, A., Kearns, C. A., Pape, T., & Thompson, F. C. (2008). Pollinating flies (Diptera): a major contribution to plant diversity and agricultural production. *Biodiversity*, *9*(1-2), 86-89. doi:10.1080/14888386.2008.9712892

Tripplehorn, C. A., & Johnson, N. F. (2005). Borror and DeLong's introduction to the study of insects. *Thomson Brooks/Cole, Belmont, California*.

Weisser, W. W., & Siemann, E. (2008). The various effects of insects on ecosystem functioning. In *Insects and ecosystem function* (pp. 3-24). Springer, Berlin, Heidelberg.

Yang, L. H., & Gratton, C. (2014). Insects as drivers of ecosystem processes. *Current Opinion in Insect Science*, *2*, 26-32.

TABLES

	500 m	1 km	1.5 km	2 km
all insects	p < 0.001*** R^2 = 0.4958 AIC = 164.2069	p = 0.0013** R^2 = 0.4355 AIC = 166.3539	p = 0.0022** R^2 = 0.3990 AIC = 167.5461	p = 0.0059** R^2 = 0.3308 AIC = 169.5862
Diptera	p = 0.0659 R^2 = 0.1373 AIC = 168.5087	p = 0.1066 R^2 = 0.0956 AIC = 169.4057	p = 0.1572 R^2 = 0.0645 AIC = 170.0482	p = 0.2516 R^2 = 0.0222 AIC = 170.8887
Hymenoptera	p = 0.0135* R^2 = 0.2682 AIC = 132.2484	p = 0.0128* R^2 = 0.2725 AIC = 132.1378	p = 0.0116* R^2 = 0.2801 AIC = 131.9375	p = 0.0084** R^2 = 0.3043 AIC = 131.2886
Coleoptera†	p = 0.0061*** R^2 = 0.3282 AIC = 44.4303	p = 0.0084** R^2 = 0.3043 AIC = 45.0958	p = 0.0067** R^2 = 0.3216 AIC = 44.6164	p = 0.0086** R^2 = 0.3031 AIC = 45.1269
Hemiptera†	p = 0.0149* R^2 = 0.2606 AIC = 28.7025	p = 0.0083* R^2 = 0.3053 AIC = 27.518	p = 0.0037** R^2 = 0.3641 AIC = 25.8376	p = 0.0027 R^2 = 0.3864 AIC = 25.1600

Table 1.1 Linear model statistics for insect order groupings across four scales of urban intensity measurement. Shaded boxes indicate statistically significant relationships, with the darkest shading representing the best model fit (lowest AIC value) for that insect grouping. Insect variables that were square root transformed to meet assumption of normality are marked with a dagger (†).
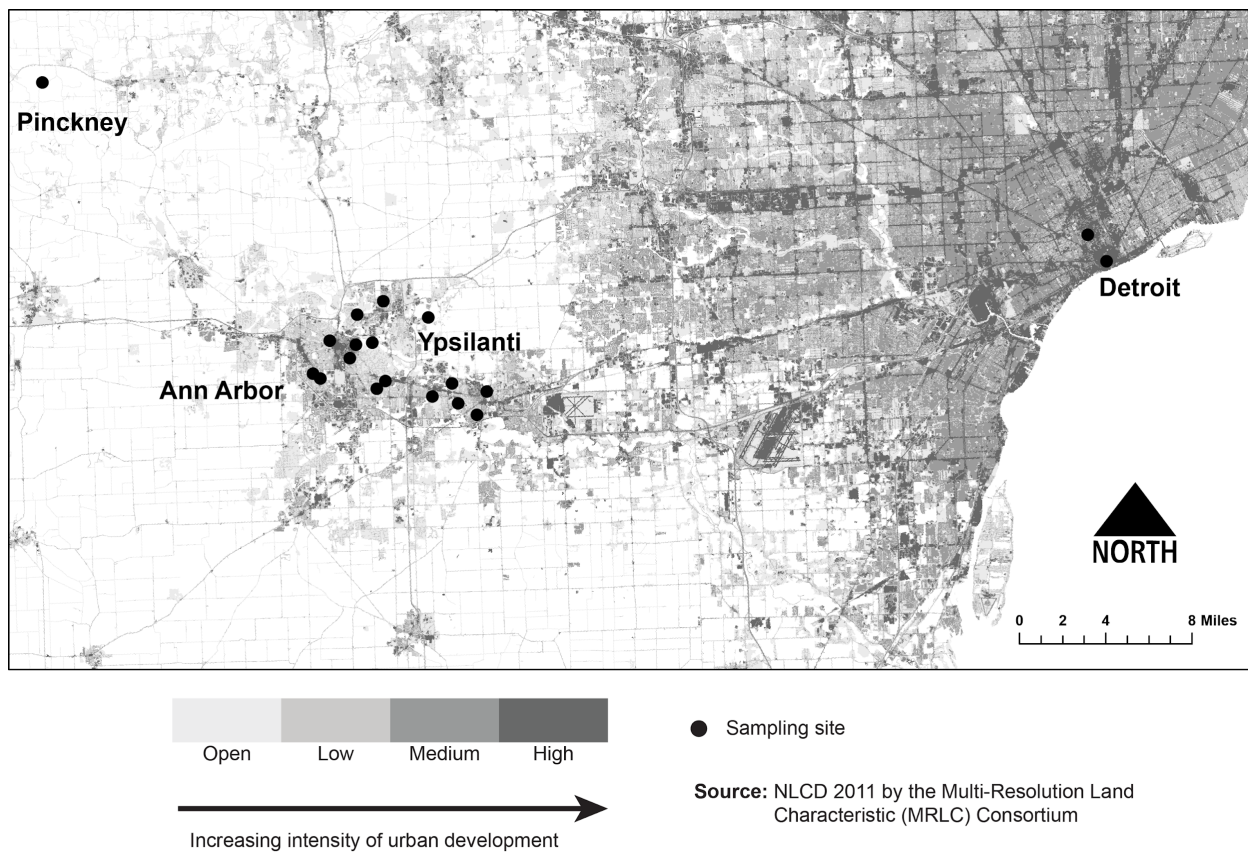
FIGURES

Figure 1.1 Map of 19 study sites in the urban-to-rural gradient in Southeast Michigan. The northwest site at Pinckney was located in the E.S. George Reserve; all other sites were located in community food gardens. Modified from M. Carolina Simao, 2014.
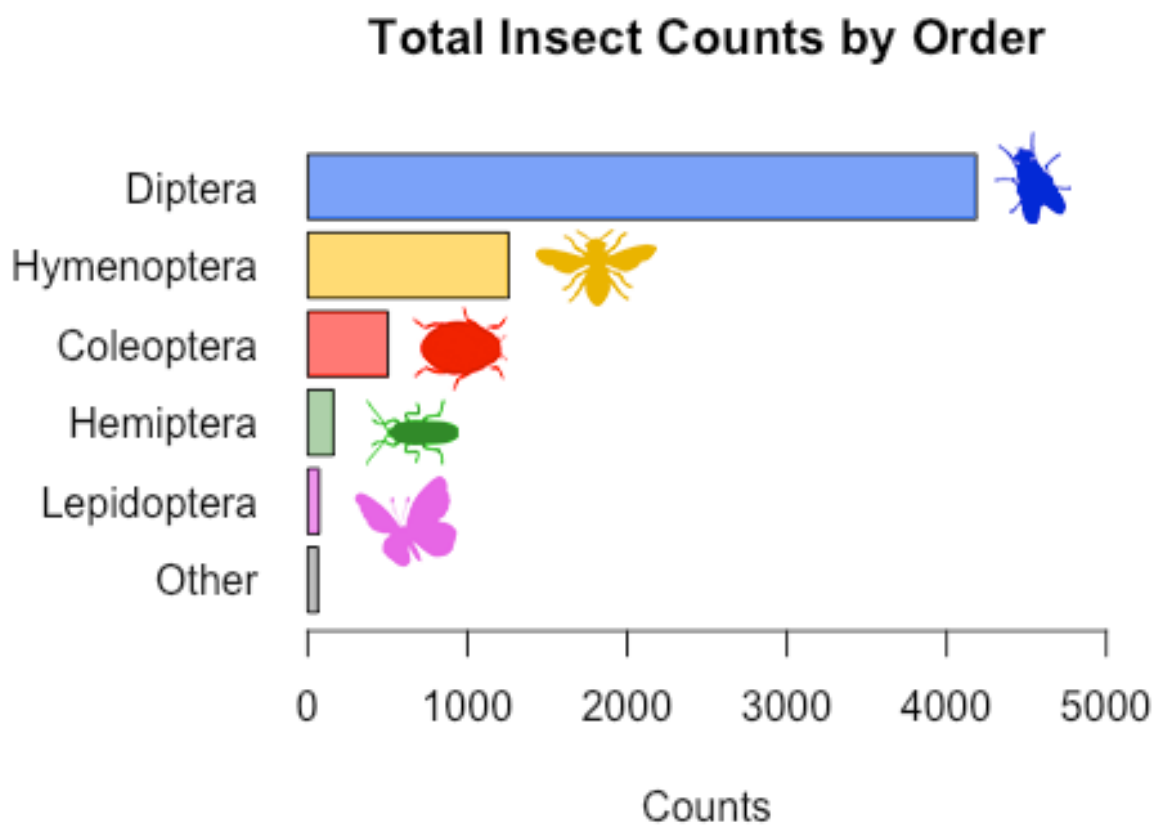
Figure 1.2 Total insect counts by order. Bar chart showing insect order counts for all sample dates and sites in the study.

Figure 1.3 Urban intensity correlates with declining insect abundance. Scatter plot of the proportion of medium- and high-level urban development (impervious surface cover >50%) within a 500 m radius of each study site versus the mean number of insects per sampling period at that site, with a best fit regression line ($p < 0.001$, $R^2 = 0.496$).

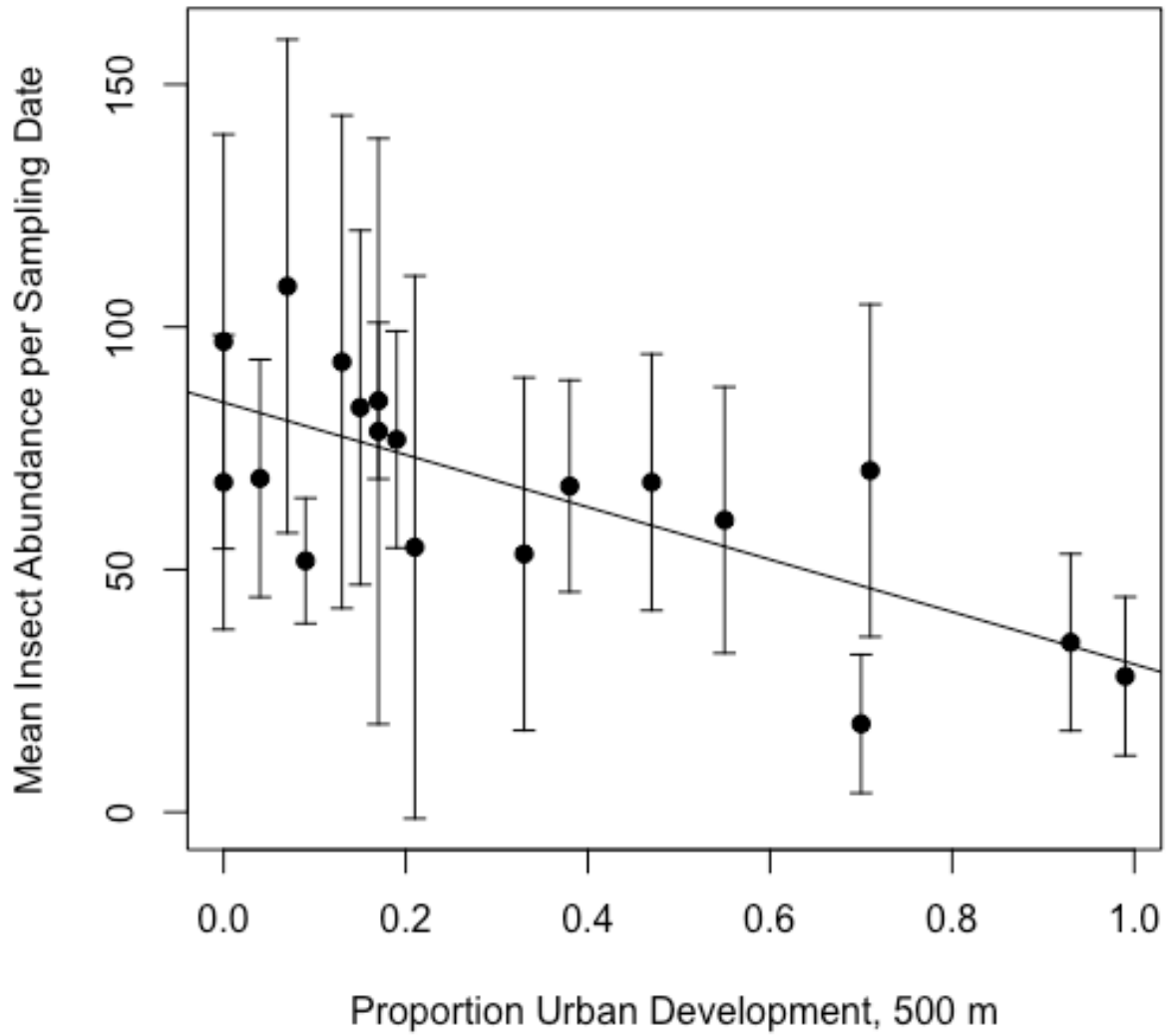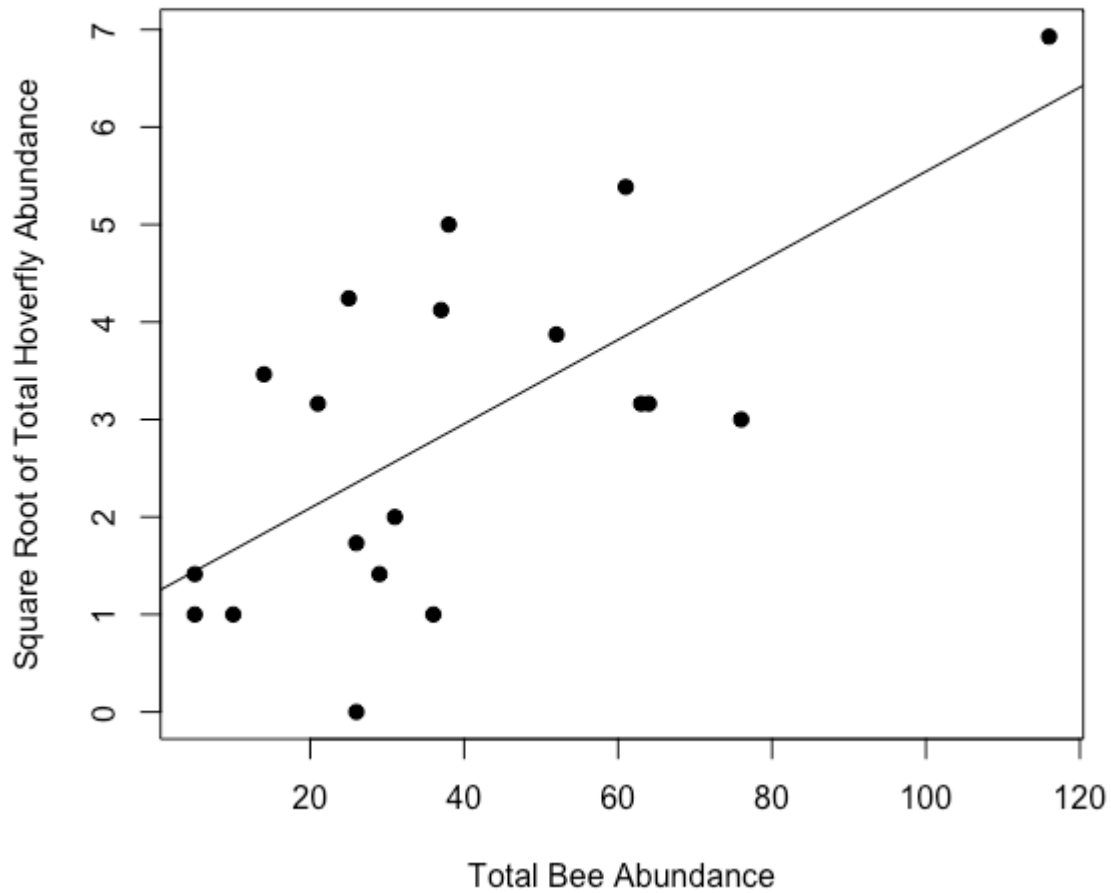**Figure 1.4** Bee abundance correlates with hoverfly abundance. Scatter plot and best fit line ($p = 0.0016$, $R^2 = 0.4209$) showing the relationship between total bee (order Hymenoptera, superfamily Apoidea) and hoverfly (order Diptera, family Syrphidae) abundance across all sampling sites and dates. Hoverfly data was square root transformed to meet the assumption of normality.

CHAPTER 2

Exploring socio-economic drivers for urban agriculture and native bees[1]

ABSTRACT

Burgeoning urban agriculture initiatives in American cities have raised questions about where gardens get established and what effects they have on local ecosystems. Socio-economic demographics may factor into patterns of garden establishment and local ecological traits, leading to changes in biotic communities. Some changes are potentially beneficial; for example, increased floral resources might benefit declining pollinator species. However, connections between socio-economic drivers and ecological effects are poorly understood. This study demonstrates how interdisciplinary methods can be used in urban ecology research by exploring effects of wealth on community gardens in southeast Michigan, with consequences for floral resources and native pollinator communities. We created socio-economic profiles of communities surrounding gardens using census land cover data to discern patterns in garden placement. A subset of these gardens was then sampled for floral resources and wild bees. Results show that while gardens are evenly dispersed across socioeconomically diverse communities, availability of undeveloped land is a greater determinants of garden location. Socio-economic variables significantly correlate with garden floral characteristics, which may have implications for urban pollinator support. Overall we present a novel combination of techniques to connect social and ecological components of sustainability.

---

[1] A version of this article first appeared in the Michigan Journal of Sustainability as "Socio-economic drivers of community garden location and quality in urban settings and potential effects on native pollinators," by Benjamin Iuliano, Alexandra Markiewicz, and Paul Glaum (2017).

INTRODUCTION

In recent years, urban agriculture has seen a marked increase in prevalence across the United States. According to estimates by the National Gardening Association, the number of urban gardeners increased by 29 percent between 2008 and 2013, and the number of households participating in community gardening increased by 200 percent to over three million during the same time period (National Gardening Association 2014). Such trends give rise to questions about how and why gardens spread, as well as where they get established and who stands to benefit most. Urban and community gardens have been touted for their ability to address a wide range of issues including food security, urban blight, community cohesion, public health, unemployment, and environmental sustainability (Santo et al. 2016). The purported sustainability potential of urban agriculture has been attributed to reduction of food miles and greenhouse gas emissions (Kulak et al. 2013), sequestration of air pollutants into plant tissue (Janhäll 2015), biodiversity preservation (Galluzzi et al. 2010), and other ecosystem services. Yet empirical impacts vary greatly with context and require more research (Santo et al. 2016).

Studies have also examined what effects gardens have on surrounding communities using a variety of indicators, and socio-economic trends are a common area of inquiry. For example, a 2012 review of urban community garden literature found 11 US studies that measured the effect of garden establishment on surrounding property values, all of which showed an increase (Guitart et al. 2012). Yet few academic studies thus far have addressed whether particular types of communities are more prone to establish and maintain community gardens than others. In one of the most comprehensive studies to date of how community gardens differ across economically and racially diverse neighborhoods, Burdine & Taylor (2018) found that urban agriculture sites

in Toledo, OH were stratified in their garden infrastructure, crop varieties, and sustainability practices, but that community participation was high across all garden types.

Such socio-demographic differences across communities might also affect the underlying environmental quality and resultant ecological structure of gardens. Links between socio-economic characteristics and the biodiversity of plants (Grove et al. 2006, Kinzig et al. 2005; Hope et al. 2003) and birds (Loss et al. 2009; Kinzig et al. 2005) in US cities have been studied at a variety of scales ranging from home gardens to neighborhoods and landscapes. Plant biodiversity in itself has been demonstrated as an important determinant of the status of animal and insect communities (Knopps et al. 1999), and this might be particularly true in urban gardens (Smith et al. 2006). For example, it has been suggested that the floral resources in urban gardens might be a valuable resource for at-risk bee communities, which are suffering global declines (Hernandez et al. 2009). Yet the broader connection between socio-economic variables and the urban bee communities remains understudied.

In order to demonstrate how researchers might go about answering such complex socio-ecological questions, we present a case study of community gardens in Ann Arbor and Ypsilanti, Michigan investigating the relationships among socio-economic demographics, community garden location, garden floral composition, and native bee abundance. Given the inherently interdisciplinary nature of this work, we employ a diverse set of methods spanning the natural and social sciences. Specifically, we ask three questions:

1. Are community gardens more prevalent in certain socio-economic conditions of urban environments? Specifically, do socio-economic variables correlate with garden placement in Ann Arbor and Ypsilanti, Michigan?

2. How do socio-economic conditions of the surrounding community correlate with community garden floral resource quality?

3. How does floral resource quality influence the native bee abundance in urban community gardens?

METHODS

Study system

Our study encompasses 30 community gardens maintained by management organizations Project Grow in Ann Arbor and Growing Hope in Ypsilanti. These sites are particularly appropriate for addressing our research questions. Both management organizations maintain readily available data about community garden locations and administer consistent management regulations across garden sites. Furthermore, the Ann Arbor/Ypsilanti area encompasses significant heterogeneity of socio-economic demographics within a limited geographic range, both making our study logistically feasible and limiting the effect of confounding variables such as divergent climates.

During the summer of 2014 we received permission to sample bee communities and floral resources across a subset of 11 of these garden sites. These gardens were located at Buhr Park, Clague Middle School, the University of Michigan Campus Farm, County Farm Park, Catholic Social Services, Eastern Michigan University, Frog Island, Greenview, Normal Park, Perry Community Garden, and West Park. All gardens ban the use of neonicotinoid pesticides, which have been linked to declining bee health (Woodcock et al. 2016; Blacquière et al. 2012), and provide guidelines to utilize organic growing practices accepted by major organic certification agencies, such as the Organic Crop Improvement Association (www.ocia.org).

GIS and socio-economic data

We used socio-economic/demographic indicators related to wealth from the US Census Bureau's 2006-2010 American Community Survey in our analyses. These included median income, percent below the poverty line, population percent with a bachelor's degree, median property value, and median age. All of the socio-economic variables considered here are associated with wealth in Ann Arbor and Ypsilanti. Median income has significant positive correlations with age, percent bachelor's degree, and property value, and a negative correlation with poverty rate. These associations correspond to our understanding of Ann Arbor/Ypsilanti demographics. Additionally, both cities are college towns, and official census rules dictate that most college students should be counted at their college addresses (census.gov/2010census).

Profiles for each site regarding each socio-economic variable were created using Geographic Information System (GIS). GIS was used to create quarter-mile (402.34 m) buffer zones around each of 30 garden sites in Ann Arbor and Ypsilanti and overlay these buffers with census tract data for each socio-economic variable. We determined the proportion of each census tract that lay within a particular buffer, multiplied the variables by this proportion, and summed the proportional values to attain the socio-economic data values associated with each garden site.

Field sampling

Native bee trapping methods for this project are consistent with those commonly found in the literature (Pardee and Philpott 2014, Fortel et al. 2012). Methods and frequency of sampling used for this study also adhered with recommendations to avoid disrupting bee communities while maintaining accurate representation in data (Banaszak et al. 2014).

Pan trapping was performed every two weeks from May to September 2014. At each site, pan traps were installed in a rectangular arrangement of eight square meters. Pan traps were coated with UV reflecting paint; two of each color blue, yellow, and white were placed at each site. Pan traps were filled with water mixed with a small amount of Dawn scentless soap, which acts to reduce the surface tension of the water to effectively trap landing insects. At sites unobscured by floral cover, the trap apparatus was left at ground level, while it was mounted on PVC pipes at sites where the ground was covered by vegetation. In addition to passive trapping, active netting was done every other collection date (once per month) so as to ensure sampling of larger species whose size or flying strength make them difficult to capture in pans. On each For each date on which netting occurred, it was done for 30 minutes both in the morning (9 AM to 12 PM) and in the afternoon (1 PM to 4 PM) within a 5-meter square centered on the trap apparatus.

To measure the amount of floral resources available to pollinators per site, floral surveys were conducted on each trap date. A logarithmic scale was used to estimate the number of flowers of each species in a 20-meter radius circle around the center of the trap setup at each garden. Species average measurements of floral geometry were used along with count estimates to calculate the geometric area of open inflorescence (hereafter referred to as "floral area") for individual species and total flowers present within the 1256.64 meters square sampling area. We evaluated garden floral quality based on mean floral area and diversity. Floral area and diversity were averaged across sampling dates per sites to account for variance in floral resources due to harvesting and sampling availability. We also evaluated specific subdivisions within floral community composition. These included the mean and proportional areas of crop, herb, ornamental, and weed species, as well as introduced, native, and invasive plant species.

Statistical analyses

Statistical analysis was completed using the statistical software R. In order to address question 1 of this study, socio-economic and demographic variables at garden locations were compared to areas without gardens. To accomplish this, buffer zones were created for randomly chosen non-garden locations. Then socio-economic variables and impervious surface coverage were measured within the buffer zones similarly to garden locations (see GIS section above). Using non-garden locations as reference points is more applicable than simply comparing garden locations to overall city averages. Using city averages would result in a fallacious comparison: garden locations would be tested against a broader landscape also containing those same garden locations. Socio-economic variables at garden locations and non-garden locations were compared directly through t-tests.

In addressing question 2, we initially set out to use socio-economic variables as predictors and floral measurements as dependent variables in multivariate regression analysis. In this process we confronted the fact that the socio-economic variables in our data exhibit "multicollinearity." In other words, the income around gardens strongly and positively correlates with property value, median age, and bachelor's degree percentage. Conversely, these variables all strongly, negatively correlate with percent in poverty. Perhaps unsurprisingly, socio-economic variables are very often collinear (Wagner 1982). Multicollinearity can lead to particularly imprecise estimates of beta coefficients in ordinary multivariate regressions. Since tests showed our data exhibits a high degree of multicollinearity, ordinary multivariate models had to be avoided. We proceeded with the analysis using single-variable linear models. Estimates of the beta coefficients of these more straightforward single–variable linear models were verified through residualization in multivariate models. Given the use of linear models, skewed variables

were log transformed to meet conditions of normality. Data was also standardized to z-scores

given that each variable was measured across significantly different ranges and scales,

streamlining comparison of effect size across variables.

RESULTS

Socio-economic conditions and garden location

Analysis of the socio-economic patterns of communities surrounding gardens revealed a

number of important quantitative and qualitative observations (Figure 2.1). Median income and

percentage of Bachelor's degrees in garden buffer areas showed no significant differences from

parts of the respective city not containing gardens. In other words, the income and education

demographics of neighborhoods surrounding gardens in Ypsilanti tended to reflect those of

Ypsilanti as a whole, and the demographics of neighborhoods surrounding gardens in Ann Arbor

was similar to Ann Arbor as a whole. Ypsilanti also showed no difference between garden and

non-garden areas for the variables of poverty and age. In Ann Arbor, however, areas surrounding

community gardens were significantly older (t = -2.766, df = 42.344, p = 0.008) and had a

significantly lower poverty rate (t = 2.603, df=51.111, p = 0.012) than non-garden areas.

Additionally, there were significant differences in garden demographics between cities, apparent

in the differential shading on the maps in Figure 2.1.

Socio-economic conditions and garden floral qualities

Across the 11 sample sites, 2086 floral abundance measurements were taken of 316

flowering plant species. Results of linear models between mean floral area and socio-economic

variables are shown in Table 2.1. Variables associated with wealth (median income, percent with

bachelor's degree, and median age), had a significant positive linear relationship with general floral resource availability in community gardens. Conversely, percent of residents below the poverty line surrounding gardens had a significant negative correlation with floral area. In general, Ann Arbor/Ypsilanti gardens in more affluent areas tended to have greater apparent floral resource availability than those in lower-income communities. Correspondingly, there was a decrease in floral resources associated with poverty.

We hypothesized that this increase in floral area associated with wealth did not constitute a proportional increase in all types of flowers. To investigate how floral composition changed with the increase in area, flower data was broken down into further descriptive categories and regressed against the same socio-economic variables as well as general floral area. Significant relationships were found among socio-economic variables and various garden flower composition variables (Figure 2.2). Some of the strongest and most consistent relationships involved mean crop area, mean introduced species area, and mean weed percent, and thus these variables will be the primary focus of our analysis.

Significant relationships were found between mean crop area in gardens and all socio-economic variables of interest in surrounding neighborhoods. As with mean floral area, wealth-associated variables were positively correlated with crop area, and poverty was negatively correlated. The same trends held true for mean introduced species floral area: introduced flowers increased with wealth and decreased with poverty. In contrast, the percentage of floral area occupied by weeds had significant negative correlations with income and a positive trend associated with poverty ($p = 0.052$) (Table 2.2).

Garden floral qualities and wild bee abundance

We collected 1,706 individual bee specimens from 112 species. The most abundant species were the bronze sweat bee (*Halictus confusus*) with 132 individuals, the common eastern bumble bee (*Bombus impatiens*) with 124 individuals, and a metallic green sweat bee species (*Agapostemon virescens*) with 116 individuals (Figure 2.3).

Neither general floral area or diversity correlated with the number of bees sampled. In fact, the relationship between floral area and bee abundance trended negative ($p = 0.057$, $R^2 = 0.270$). This trend might be driven by the significant negative interaction between introduced species floral area and native bee abundance. On the other hand, there was a nearly significant positive trend ($p = 0.052$, $R^2 = 0.29$) between percent weeds and bee abundance; the number of bees tended to increase as a greater proportion of garden floral area was occupied by weeds. Finally, there was no significant relationship between crop area and bee abundance.

DISCUSSION

In summary, the analysis of how socio-economic demographics correspond to community garden locations showed significant differences in Ann Arbor between median age and poverty rates in areas surrounding gardens versus non-garden areas. Such differences were not consistent in Ypsilanti, where gardens were more evenly dispersed across neighborhoods. On the other hand, socio-economic factors were closely correlated with garden floral composition. Furthermore, while urban community gardens can be successful in their ability to support pollinator communities, intuitive garden traits within local control might not explain differences in this ability between gardens. In Ann Arbor and Ypsilanti community gardens, we find wealth is correlated with high floral areas, but that does not necessarily translate to more abundant bee

communities (Figure 2.4)

In general, the socio-economic makeup of neighborhoods surrounding community gardens tended to reflect the socio-economic makeup of their respective cities. In Ypsilanti, there was no significant difference for any socio-economic variables. In Ann Arbor there was no difference for income, education, or property value, while there were significant differences for poverty and age. The city's much larger student population might explain these differences in Ann Arbor. Students who are not generally present during the peak growing months tend to live in the denser, centrally located neighborhoods. Ann Arbor residents participating in community gardening are likely to be older, permanent inhabitants living near lower-density residential neighborhoods that contain more open space for gardens. The large student population would thus decrease the median age in areas without gardens relative to those areas where gardens are located. Similarly, the presence of off-campus college students has been shown to skew census poverty rates upward (Bishaw 2013). Thus, we would expect non-garden areas occupied by students to have higher poverty rates than garden areas.

The gardens in Ann Arbor and Ypsilanti spanned diverse communities, demonstrating that gardens are established and persist regardless of a community's characteristics. However, structural barriers such as the accessibility of land are likely to pose a bigger challenge, no matter how strong the environmental, ecological, or community-oriented incentives to do so may be (Mubvami et al. 2003). Vacant land may be unavailable or the cost of land may be insurmountable for an individual or community motivated to develop a community garden. Therefore, strengthening or expanding urban agriculture will require the attention of city planners and policy makers. Urban agriculture-friendly zoning or land use designations, along with initiatives such as providing gardens with city water access or allowing communities to

develop gardens in public parks, are some ways in which municipal stakeholders can, and may need to, support community gardens.

Beyond garden location, characteristics of community garden floral quality are significantly correlated with socio-economic variables, indicating that even factors ostensibly within the purview of local- or individual-level decision-making might be influenced by broader demographic trends. Our analysis found that variables associated with wealth, such as low poverty and high income, educational attainment, and age tended to positively correlate with floral area in Ann Arbor and Ypsilanti community gardens. This increase in general floral area was primarily driven by increases in crops and introduced plant species.

Furthermore, there was a negative association between income and percentage of floral area occupied by weeds. Although we do not have qualitative data such as gardener interviews to suggest a causal mechanism for these relationships, we can propose potential explanatory hypotheses. Gardens in wealthier areas, with higher incomes and lower poverty levels, plausibly attract wealthier gardeners who are better resourced and able to manage garden plots more intensively. This could explain the significant increases in crop area (i.e. wealthier gardeners purchase and plant more crops) and corresponding increase in general floral area, as well as the decrease in percent weeds (i.e. wealthier gardeners have more resources, time, and/or concern for weed management). An analogous "luxury effect" has been found in other research on vegetation in urban areas, such as a study which demonstrated socio-economic drivers of plant diversity around Phoenix, AZ (Hope et al. 2003).

One might presume that these characteristics make gardens in wealthier areas "higher quality." In particular, the significantly higher floral area in affluent gardens seems likely to be more attractive and beneficial to wild bees. Yet our findings do not support this assumption. In

fact, increased floral area had a nearly significant negative trend with bee abundance in the 11 community gardens studied. Counter to what one might expect, we found that fewer bees were present as gardens had more flowers.

Parsing general floral area into particular categories might help explain this counterintuitive result. Specifically, increased floral abundance is driven primarily by addition of crops and introduced plant species, and the latter is also negatively correlated with bee abundance. The one floral variable that had a positive trend on bee abundance was the proportion of the floral area occupied by weed plants ("percent weeds"). The value of weeds to pollinators has been well documented, summarized in a recent FAO review (Altieri et al. 2015). Bees tend to be attracted to particular flower types based on nutrient quality, such as pollen protein content (Hanley et al. 2008), and non-crop flowers provide bees with the requisite diet diversity for healthy development (Alaux et al. 2010; Schmitt et al. 1995). It is therefore plausible that increased proportion of crop and/or introduced flowers relative to weeds in gardens "dampens the signal" of weed flowers that are attractive to wild bees, though more work should be done to test this mechanistically. Ultimately our results suggest that while urban community gardens are capable of supporting bee communities, moving beyond a paradigm of simple floral additions could be important. The composition of the floral resources in gardens matters, and urban gardeners should consider incorporating more non-crop, wild flowers along with crops in order to support bee health.

Our study design necessarily entails limitations that should be accounted for when extrapolating results. First, using census data conducted at such a broad scale is not a perfect on-the-ground measurement of the socio-economic reality of the communities surrounding urban gardens or the wealth status of gardeners themselves—although the findings of Burdine &

Taylor (2017) suggest that gardener demographics tend to align with the demographics of surrounding neighborhoods. Flower and bee sampling only took place over one growing season, and thus cannot account for longer-term temporal variation in community abundance and composition. Furthermore, our analyses did not account for other garden qualities besides floral resources such as nesting habitat availability. Finally, definitive causal relationships can only be inferred due to both the observational nature of our study and the small sample size in a specific local context. Yet our results do suggest future research directions for investigating similar trends in other urban contexts as well as the mechanistic interactions among socio-economic drivers, garden location/quality, and pollinator communities. Perhaps most importantly, the interdisciplinary methodology employed here exemplifies a novel and potentially useful tool for addressing complex questions associated with social and ecological sustainability in cities. If urban gardens are anticipated to have wide-reaching effects that cut across socio-cultural, economic, public health, and environmental dimensions, research methods to study these effects must be similarly broad and diverse. Our study constitutes an illustrative example of one such approach. Incorporating similar methods into future research questions could deepen understanding of the broader relationships between socio-economic characteristics, land use, and the prevalence of community gardens, and identify the best associated policy recommendations that would foster the expansion of urban agriculture and improve conditions for biotic communities.

WORKS CITED

Alaux, C., Ducloz, F., Crauser, D., & Le Conte, Y. (2010). Diet effects on honeybee immunocompetence. *Biology letters*, 6(4). doi:10.1098/rsbl.2009.0986.

Altieri, M. A., Nicholls, C. I., Gillespie, M., Waterhouse, B., Wratten, S., Gbèhounou, G., & Gemmill-Herren, B. (2015). Crops, weeds and pollinators: understanding ecological interactions for better management. *Food and Agriculture Organization of the United Nations*. http://www.fao.org/3/a-i3821e.pdf.

Banaszak, J., Banaszak-Cibicka, W., & Szefer, P. (2014). Guidelines on sampling intensity of bees (Hymenoptera: Apoidea: Apiformes). *Journal of insect conservation*, *18*(4), 651-656. doi:10.1007/s10841-014-9671-3.

Bishaw, A. (2013). Examining the effect of off-campus college students on poverty rates. *United States Census Bureau, Social, Economic, and Housing Statistic Division, Poverty Statistics Branch*. http://www.census.gov/content/dam/Census/library/working-papers/2013/acs/2013_Bishaw_01.pdf

Blacquiere, T., Smagghe, G., Van Gestel, C. A., & Mommaerts, V. (2012). Neonicotinoids in bees: a review on concentrations, side-effects and risk assessment. *Ecotoxicology*, *21*(4), 973-992. doi: 10.1007/s10646-012-0863-x

Burdine, J. D., & Taylor, D. E. (2018). Neighbourhood characteristics and urban gardens in the Toledo metropolitan area: staffing and voluntarism, food production, infrastructure, and sustainability practices. *Local Environment*, 23(2), 198-219. doi:10.1080/13549839.2017.1397614

Fortel, L., Henry, M., Guilbaud, L., Guirao, A. L., Kuhlmann, M., Mouret, H., ... & Vaissière, B. E. (2014). Decreasing abundance, increasing diversity and changing structure of the wild bee community (Hymenoptera: Anthophila) along an urbanization gradient. *PloS one*, *9*(8), e104679. doi:10.1371/journal.pone.0104679.

Galluzzi, G., Eyzaguirre, P., & Negri, V. (2010). Home gardens: neglected hotspots of agro-biodiversity and cultural diversity. *Biodiversity and conservation*, *19*(13), 3635-3654. doi:10.1007/s10531-010-9919-5.

Grove, J. M., Troy, A. R., O'Neil-Dunne, J. P., Burch, W. R., Cadenasso, M. L., & Pickett, S. T. A. (2006). Characterization of households and its implications for the vegetation of urban ecosystems. *Ecosystems*, 9(4), 578-597. doi:10.1007/s10021-006-0116-z.

Guitart, D., Pickering, C., & Byrne, J. (2012). Past results and future directions in urban community gardens research. *Urban forestry & urban greening*, *11*(4), 364-373. doi:10.1016/j.ufug.2012.06.007.

Hanley, M. E., Franco, M., Pichon, S., Darvill, B., & Goulson, D. (2008). Breeding system, pollinator choice and variation in pollen quality in British herbaceous plants. *Functional ecology*, *22*(4), 592-598. doi:10.1111/j.1365-2435.2008.01415.x.

Hernandez, J. L., Frankie, G. W., & Thorp, R. W. (2009). Ecology of urban bees: a review of current knowledge and directions for future study. *Cities and the environment (CATE)*, *2*(1), 3. http://digitalcommons.lmu.edu/cgi/viewcontent.cgi?article=1036&context=cate

Hope, D., Gries, C., Zhu, W., Fagan, W. F., Redman, C. L., Grimm, N. B., Nelson, A.L., Martin, C., & Kinzig, A. (2003). Socioeconomics drive urban plant diversity. *Proceedings of the national academy of sciences*, *100*(15), 8788-8792. doi:10.1007/978-0-387-73412-5_21.

Janhäll, S. (2015). Review on urban vegetation and particle air pollution–Deposition and dispersion. *Atmospheric environment*, *105*, 130-137. doi:10.1016/j.atmosenv.2015.01.052.

Kinzig, A., Warren, P., Martin, C., Hope, D., & Katti, M. (2005). The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. *Ecology and society*, *10*(1). doi:10.5751/ES-01264-100123.

Knops, J. M., Tilman, D., Haddad, N. M., Naeem, S., Mitchell, C. E., Haarstad, J., Ritchie, M.E., Howe, K.M., Reich, P.B., Siemann, E., & Groth, J. (1999). Effects of plant species richness on invasion dynamics, disease outbreaks, insect abundances and diversity. *Ecology Letters*, 2(5), 286-293. doi: 10.1046/j.1461-0248.1999.00083.x

Kulak, M., Graves, A., & Chatterton, J. (2013). Reducing greenhouse gas emissions with urban agriculture: a life cycle assessment perspective. *Landscape and urban planning*, *111*, 68-78. doi:10.1016/j.landurbplan.2012.11.007.

Loss, S. R., Ruiz, M. O., & Brawn, J. D. (2009). Relationships between avian diversity, neighborhood age, income, and environmental characteristics of an urban landscape. *Biological Conservation*, *142*(11), 2578-2585. doi:10.1016/j.biocon.2009.06.004

Mubvami, T., Mushamba, S., & van Veenhuizen, R. (2003). Availability, access and usability of land for urban agriculture. *Urban agriculture magazine*, *11*, 1-3. http://www.ruaf.org/node/1677

National Gardening Association (2014). "Garden to Table: A 5 Year Look at Food Gardening in America" *National Gardening Association*. https://garden.org/special/pdf/2014-NGA-Garden-to-Table.pdf.

Pardee, G. L., & Philpott, S. M. (2014). Native plants are the bee's knees: local and landscape predictors of bee richness and abundance in backyard gardens. *Urban ecosystems*, *17*(3), 641-659.doi:10.1007/s11252-014-0349-0

Santo, R., Palmer, A., & Kim, B. (2016). Vacant Lots to Vibrant Plots: A review of the benefits and limitations of urban agriculture. *Center for a Livable Future, John Hopkins University*. http://www.jhsph.edu/research/centers-and-institutes/johns-hopkins-center-for-a-livable-future/_pdf/research/clf_reports/urban-ag-literature-review.pdf.

Schmidt, L. S., Schmidt, J. O., Rao, H., Wang, W., & Xu, L. (1995). Feeding preference and survival of young worker honey bees (Hymenoptera: Apidae) fed rape, sesame, and sunflower pollen. *Journal of economic entomology*, *88*(6), 1591-1595. doi:10.1093/jee/88.6.1591.

Smith, R. M., Warren, P. H., Thompson, K., & Gaston, K. J. (2006). Urban domestic gardens (VI): environmental correlates of invertebrate species richness. Biodiversity & Conservation, 15(8), 2415-2438. doi:10.1007/s10531-004-5014-0.

Wagner, C. H. (1982). Simpson's paradox in real life. *The American statistician*, *36*(1), 46-48. doi:10.2307/2684093.

Woodcock, B. A., Isaac, N. J., Bullock, J. M., Roy, D. B., Garthwaite, D. G., Crowe, A., & Pywell, R. F. (2016). Impacts of neonicotinoid use on long-term population changes in wild bees in England. *Nature communications*, *7*, 12459. doi:10.1038/ncomms12459.

TABLES

Predictor Variables	ln(Mean Floral Area)			
	β	$F_{1,9}$	p-value	R^2
% in poverty	- 0.73 ± 0.22	10.33	0.011*	0.48
Median income	+ 0.68 ± 0.23	7.685	0.022*	0.40
% with bachelors	+ 0.69 ± 0.24	8.043	0.020*	0.41
Median age	+ 0.81 ± 0.19	17.28	0.003**	0.62
Property value	+ 0.53 ± 0.28	3.564	0.092	0.20

Table 2.1 Relationships between socio-economic predictors and mean floral area. Values are from simple linear regression models. Significance of model fits is given by p-values: * < 0.05, ** < 0.01.
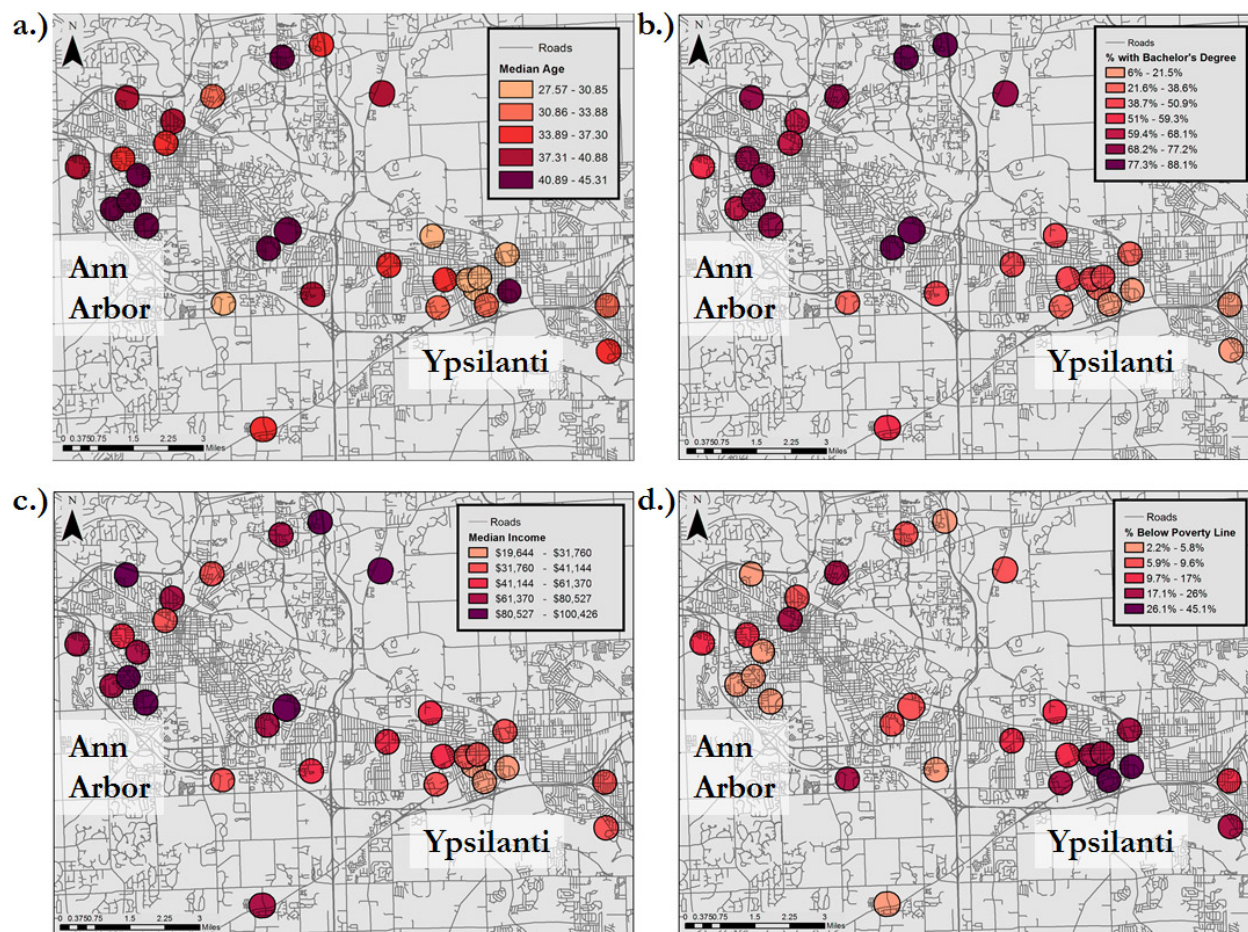
FIGURES

Figure 2.1 Socio-economic demographics in garden buffer zones in Ann Arbor and Ypsilanti, MI. Circles represent buffer zones with garden at center. a) Median age b) Percent of bachelor's degrees c) Median income d) Percent below the poverty line. Created by Alexandra Markiewicz.
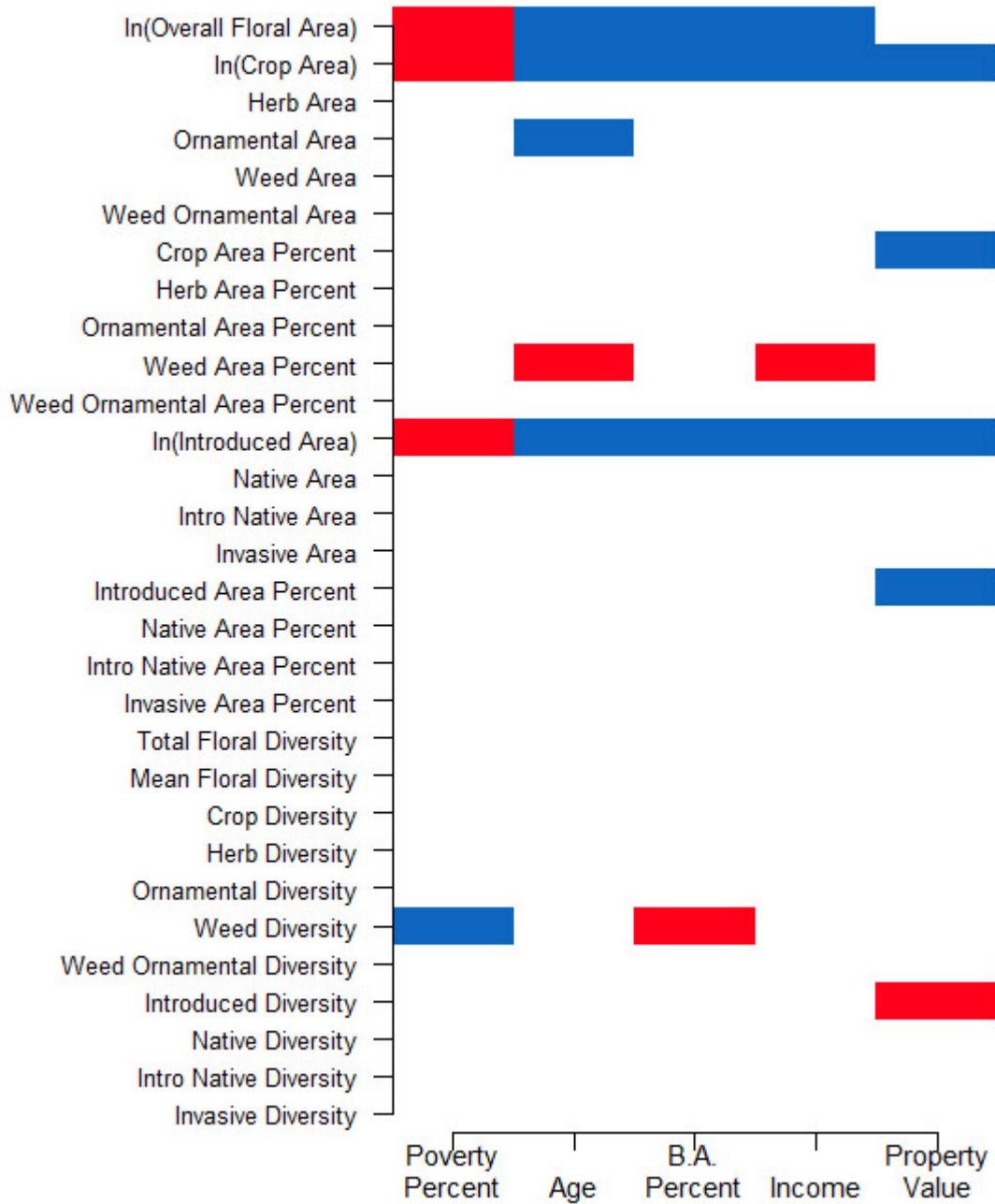
53

Figure 2.2 Heat map of beta estimates from statistically significant relationships between socio-economic and floral variables. Significance determined at the $\alpha = 0.05$ level. Colors represent the sign value of the beta estimates of linear regressions. Blue colors represent positive relationships; red colors represent negative relationships. Created by Paul Glaum.
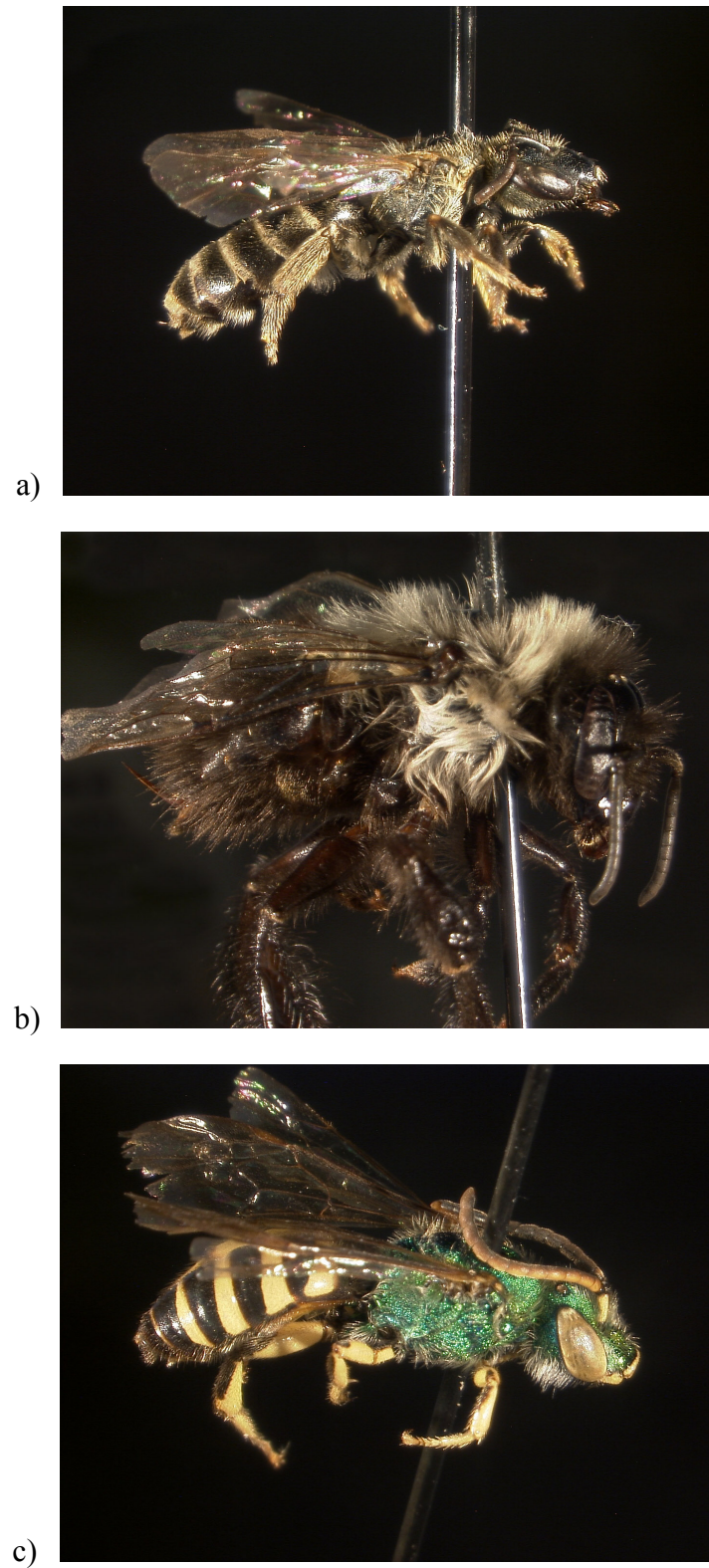
Figure 2.3 Photographs of the three most common bee species in our sample: a) *Halictus confusus* b) *Bombus impatiens* c) *Agapostemon virescens*.
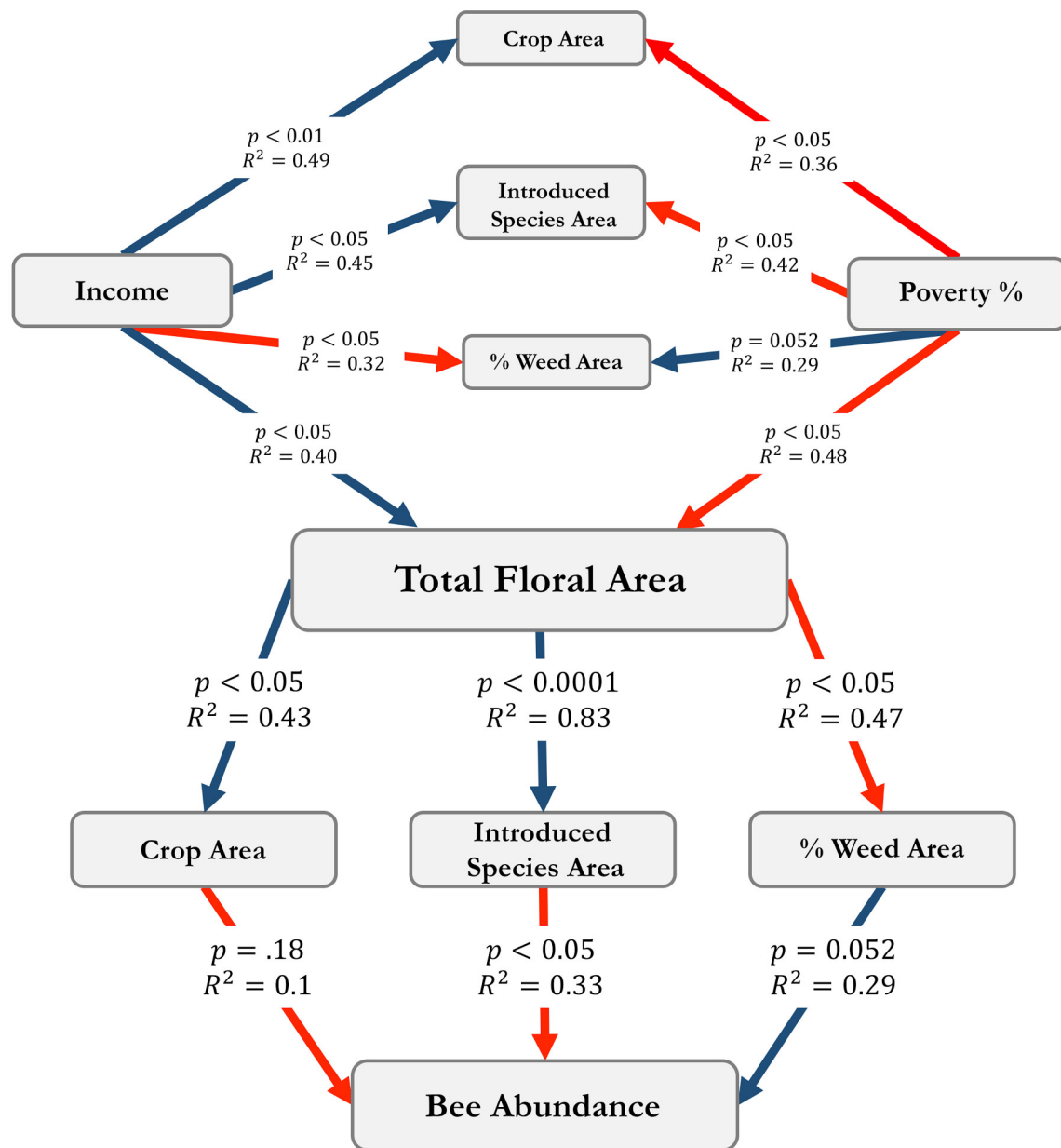
Figure 2.4 Conceptual flow diagram of the relationships among income, poverty, garden floral qualities, and bee abundance. Lines represent linear correlations, with arrows pointing toward the dependent variable. Line colors indicate the direction of the relationship (blue for positive or red for negative), p-value to indicate significance ($\alpha = 0.05$), and adjusted R^2 to show the model fit.

CONCLUSION

The severe consequences of human land use for the abundance and diversity of life are patent, and the challenge that humanity faces lies in identifying the particular forces driving negative environmental change and crafting interventions to reverse them. The forces in question span political, social, economic, and ecological realms, and are often co-productive with one another. This work has explored some of these forces and their relationships in the context of urban agroecosystems. Chapter 1 demonstrated how urbanization contributes to declining insect abundance, and suggested the importance of focusing on the relationships between the specific ecological roles of insect taxa and environmental conditions across scales. The abundance of bees, wasps and ants; beetles; bugs; and insects in aggregate declined as the amount of impervious surface surrounding gardens increased. However, groups responded differently depending on the scales of impervious surface measurement, potentially attributable to distinct life histories. Both insect abundance and impervious surface were also correlated with the floral area of weeds, suggesting potential synergies between local- and landscape- drivers of urban insect population trends. These results demonstrate that particular management practices and landscape patterns are not expected to affect all insects in the same way, let alone other classes of fauna. They suggest that there will be difficult decisions to make in designing and managing urban agricultural systems depending on conservation priorities, and stakeholders will need to be clear about what desirable outcomes look like.

Chapter 2 proposed that ecological conditions do not exist in a vacuum, but are often a product of socio-economic context. The findings of our study agreed with other literature showing that gardens can exist and persist in both high- and low-income communities, but we

also found a correlation between the wealth and age status of garden neighborhoods and the relative number of crop and non-crop ("weed") flowers in gardens. Given the importance of non-crop flowers to pollinators specifically and insects generally, this illustrates an empirical mechanism for linking socio-economic and ecological forces in urban gardens. Works from the fields of political ecology and ecological design have shown that aesthetic preferences based in class and culture influence weed management in urban, suburban, and agricultural landscapes (Robbins 2012, Nassauer 1995). Similar biases also present a particular challenges for insect conservation in human-dominated landscapes, because "the sad fact is that most people don't like insects" (Hunter & Hunter 2008). These examples makes clear that it is not just the physical infrastructure of urban agriculture that matters for its conservation potential, but also the human values surrounding it.

Going forward, I echo calls for urban ecology to "scale up" (Goddard et al. 2010) and move beyond "low-hanging fruit" (McDonnell & Hahs 2013), and I contend that these calls can be applied specifically to urban agroecology and insect conservation. On the research side, this could entail more explicit consideration of landscape patterns in the matrix around urban farms and gardens, studying garden/farm assemblages across multiple cities within single studies, and crafting more applied, mechanistic research questions relevant to agricultural ecosystem services. On the practical side, municipal stakeholders should apply current ecological knowledge to the design and management of agricultural systems in cities. For example, cities could incentivized gardens to cluster in ways that facilitate desirable metapopulation dynamics (Colding 2006) or to incorporate agroforestry practices to bolster structural complexity, creating a high-quality urban matrix for urban fauna. Ultimately ecological land management for insect conservation in cities must be compatible with public sensibilities if it is to be sustainable in the long term, and

integrating aesthetically pleasing design with entomological education is likely to be valuable strategy for achieving this (Hunter & Hunter 2008). Urban agroecosystems might provide a potent locale for such intervention due to the attractive cultural power of food.

It is important to keep in mind that the *type* of agriculture in question is pivotal for matrix quality and conservation potential (Perfecto et al. 2009). Exporting an industrial agriculture model from rural to urban contexts, as has been attempted in cities including Detroit (Giorda 2012), is unlikely to provide the social and environmental benefits of urban agroecology. Some scholars have used the Marxian concept of "metabolic rift" to theorize the potential for urban agriculture, and urban agroecology specifically, to address socio-ecological disasters wrought by capitalist industrialization. "Metabolic rift" describes the alienation of humans from nature under capitalism, and the ecological crises this produces (Foster 1999). Sage et al. (2016) argue that urban agroecology can help "mend" the metabolic rift by reconnecting geographies of production and consumption, as well as creating a radical alternative urbanism that simultaneously rejects technocratic fixes and pre-urban romanticism. Similarly, McClintock (2010) sees opportunities for urban agriculture advocates to seize upon the "fault lines and fractures" created by capitalist agriculture "to transform the agri-food system into one more equitable, healthy, and ecologically sustainable."

In pursuit of such a lofty goal, urban agroecologists should continue to push disciplinary boundaries both within and without academia. Like conservation, urban agriculture is an inherently "interdisciplinary" undertaking—gardens are "socio-ecological constructs" (Goddard et al. 2010) that operate at the interface of human and non-human nature—and researchers must work across ecology, agronomy, urban planning, landscape architecture, sociology, and more if it is to play a role in shaping better food systems. Academics must be in conversation with

municipal stakeholders, from garden managers to city planners, to leverage what is already

known as well as answer new questions about conservation and ecosystem service provisioning

in urban agricultural systems.

WORKS CITED

Colding, J. (2007). 'Ecological land-use complementation' for building resilience in urban ecosystems. *Landscape and urban planning*, *81*(1-2), 46-55. doi:10.1016/j.landurbplan.2006.10.016

Gardiner, M. M., Prajzner, S. P., Burkman, C. E., Albro, S., & Grewal, P. S. (2014). Vacant land conversion to community gardens: influences on generalist arthropod predators and biocontrol services in urban greenspaces. *Urban ecosystems*, *17*(1), 101-122. doi:10.1007/s11252-013-0303-6

Goddard, M. A., Dougill, A. J., & Benton, T. G. (2010). Scaling up from gardens: biodiversity conservation in urban environments. *Trends in ecology & evolution*, *25*(2), 90-98. doi:doi.org/10.1016/j.tree.2009.07.016

Hunter, M. R., & Hunter, M. D. (2008). Designing for conservation of insects in the built environment. *Insect Conservation and diversity*, *1*(4), 189-196. doi:10.1111/j.1752-4598.2008.00024.x

McClintock, N. (2010). Why farm the city? Theorizing urban agriculture through a lens of metabolic rift. *Cambridge journal of regions, economy and society*, *3*(2), 191-207. doi:10.1093/cjres/rsq005

McDonnell, M. J., & Hahs, A. K. (2013). The future of urban biodiversity research: moving beyond the 'low-hanging fruit'. *Urban Ecosystems*, *16*(3), 397-409. doi:10.1007/s11252-013-0315-2

Nassauer, J. I. (1995). Messy ecosystems, orderly frames. *Landscape journal*, *14*(2), 161-170. doi:10.3368/lj.14.2.161

Perfecto, I., Vandermeer, J., & Wright, A. (2009). *Nature's matrix: linking agriculture, conservation and food sovereignty*. Routledge.

Robbins, P. (2012). Lawn people: *How grasses, weeds, and chemicals make us who we are*. Temple University Press.

Sage, C., Dehaene, M., & Sage, C. (2016). Mending the metabolic rift: Placing the 'urban' in urban agriculture. In *Urban agriculture europe* (pp. 174-177). Berlin: Jovis.