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A bug eat bug world : does urbanization decrease survivorship of pollinators.

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A bug eat bug world: Does urbanization decrease survivorship of
pollinators?

By:

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Submitted in partial fulfillment of the requirement for

Graduation *summa cum laude*

And

For Graduation with Honors from the Department of Biology

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Abstract

Human impact on the environment is responsible for the sixth mass extinction on Earth as well as rapid population declines in many taxa. Land use change, via increasing urbanization and intensity of agricultural practices, is the most important aspect of human impact on the environment, and it is associated with population decline and extinction in several taxa, but especially in pollinator species. Pollinators, which are essential for crop production, are experiencing declines as a result of the loss of habitat and host plants associated with urbanization. Urbanization negatively affects pollinators, but in the wake of lost naturalized habitat, some researchers are turning to urban native plant gardens as conservation spaces for pollinator populations. The current study examines whether urban native plant gardens have higher rates of predation on pollinator species as compared to naturalized ecosystems. To determine this, we chose 24 garden sites in Louisville, KY along an urban-suburban gradient. We placed clay caterpillar models on the plants within these gardens and as the models were attacked, we categorized the resulting damage according to predator taxon. Although amount of predation did not vary with urbanization, the proportion of damage by each predator taxon did: wasp predation rates increased with increasing urbanization, while vertebrate and spider predation rates decreased with increasing urbanization. Therefore, with proper maintenance and management, urban gardens may serve as suitable conservation space for common pollinator species.

Introduction

Mass extinction crisis

Earth is in the midst of its sixth mass extinction, and the cause of this species decline is directly attributed to the impact of humans on the environment (Ceballos et al., 2015). The effects of this crisis have been best studied among vertebrates, a group within which human-caused environmental changes have broadly affected several taxonomic groups, including reptiles, mammals, and amphibians. Even with conservative estimates, over the past century, the average rate of loss of vertebrate species is up to 100 times higher than the expected background rate of loss (Ceballos et al., 2015), even for species that are not currently threatened (Ceballos, Ehrlich, & Dirzo, 2017; Ceballos et al., 2015). This increased rate of loss, as a result of human impact, caused vertebrate species to become extinct within one hundred years, when it would have taken from 800 to 10,000 years to become extinct (depending on the taxon) according to the background rate of loss (Ceballos et al., 2015). Many extant vertebrate species are facing rapid population declines. One study found that 32% of examined vertebrate species have suffered reduced population size and range (Ceballos, Ehrlich, & Dirzo, 2015). Although the data regarding recent reptilian population declines is confounded and understudied (Alroy, 2015), there is detailed data regarding mammal and amphibian population declines. For example, one study found that of the mammal species with detailed data, all have lost at least 30% of their geographical range, and over 40% of these species have encountered acute population reduction, as signified by over 80% reduction in geographic range (Ceballos, Ehrlich, & Dirzo, 2017). One-third or more of amphibian species are currently threatened with extinction, a proportion which will likely increase due to the small geographic range that many tropical species occupy (Wake & Vrendenburg, 2008). As the human impact – primarily through habitat destruction and climate

change – intensifies, there are sure to be major repercussions for several vertebrate taxa, especially on narrowly adapted species and those with small geographic ranges (Wake & Vrendenburg, 2008).

The data collected on species extinction and endangerment are overwhelmingly skewed towards vertebrate taxa, which are also beneficiaries of higher proportions of conservation actions than invertebrates. This discrepancy in research emphasis may signify that only a fraction of the species to become extinct as a result of the sixth mass extinction crisis have been identified: 0.04% of modern species are confirmed to have recently become extinct, but a study conducted by Régnier et al. (2015) estimates that this percentage could actually be as high as 7% of the species currently on Earth. Invertebrates are seldom evaluated for population decline, with the exception of “charismatic” groups (e.g., butterflies, dragonflies, snails) that happen to be well-studied (Régnier et al., 2015). Although there is a paucity of information on invertebrate conservation as a whole, there is a considerable amount of information collected on the human impact on one group of invertebrate keystone species: pollinators.

Human Impact on Pollinator Species

Many pollinator species have been negatively impacted by human intervention, which may be indicative of a more extensive but unrecognized biodiversity crisis in insects (Bates et al., 2014). Prominent examples of affected groups include butterflies, moths, and native bees. Van Dyck et al. (2016) investigated multiple butterfly species’ populations from the early 1990s to mid-2000s in Northern Europe and discovered that over half of the common and widespread species experienced severe drops in both abundance and distribution. For a more specific instance, the monarch butterfly (*Danaus plexippus*) population has dropped by roughly 80% in less than fifteen years (Semmens et al., 2016), hitting record low numbers in the winter of 2013-

2014 (Cutting & Talamer, 2015). The probability of this species becoming quasi-extinct within the next few decades has been estimated to be almost 60%; halving this risk requires a five-fold increase in the current population size as well as active efforts to reverse the declining population trend (Semmens et al., 2016). Several common and widespread pollinating moth species of highly-developed Northern Europe have also experienced sharp declines in population within the past few decades (Bates et al., 2014). Yet another sect of pollinators to experience deterioration in abundance and diversity are native bees. By comparing historical data with current data gathered over three years on four species of bumble bee, one study demonstrated that not only have the surveyed geographical ranges in the United states dwindled by 23-87%, but also the relative abundances of those species have deteriorated by up to 96% (Cameron et al., 2011). Therefore, it is clear that the detrimental effects of human populations are not localized to a small segment of pollinators.

Land Use Change

The primary and most prominent suspected causes for the population decline, decreased range of location, and extinction experienced by pollinator species are the side effects of human interference with the environment – factors resulting from either urbanization or horticultural efforts, or a combination of the two (Fortel et al., 2014; Potts et al., 2016). Such factors can be characterized by the ways in which they affect pollinator species, such as: destruction of habitat and/or loss of host plant, interruption of pollination networks, and targeted destruction of native plant species in favor of nonnative plant species. Examining the factors that affect pollinator populations is crucial, since the demand among crops and native plants for pollinator species is increasing at a great rate while pollinator populations are generally declining (Potts et al., 2010). The fact that human populations are primarily responsible for the rates at which pollinator

populations are plummeting is of major concern because the majority of plants, including crops, are dependent on pollinators and such a reduction entails a potentially significant drop in food supply (Kluzer & Peduzzi, 2007).

The loss of pollinator habitat and host plants are the most crucial factors affecting pollinator population declines (Carvalho et al., 2010). This loss can be attributed in part to land use change, both via urbanization and selective horticulture, through increasing impervious surface and weeding out native host plants, respectively. For example, local butterfly diversity is negatively correlated with rising human populations in the Washington D.C. area (Di Mauro, Dietz, & Rockwood, 2007). Impervious surface has become tantamount to human presence as the natural landscape is paved over to make room for structures such as roads or sidewalks (Arnold & Gibbons, 1996), and selective horticulture typically entails a decreased proportion of the native host plants for pollinator species in favor of crops and/or other flowering nonnative plants (Jeschke et al., 2014); both of these factors result in native plant population decline by diminishing the size of habitat and limiting the biodiversity of native plant species within, which may damage naturally-occurring ecosystems (Jeschke et al., 2014). For example, the monarch butterfly's dramatic population declines are partially due to loss of its host plant, milkweed, on breeding grounds (Flockhart et al., 2015); the drop in the abundance of milkweed plants is highly correlated with the increased use of herbicide-resistant crops (Semmens et al., 2016) as well as loss of territory that the milkweed plants can occupy, and therefore, is largely a result of both direct and indirect human intervention with the natural environment via urbanization and horticulture (Flockhart et al., 2015).

Urbanization may also factor into the interruption and alteration of mutualistic pollination networks. For example, light pollution has negative consequences for plant reproductive success

(Knop et al., 2017). According to a study conducted by Knop et al. (2017), light pollution reduced the visits to plants by nocturnal pollinators by 62% when they were illuminated as compared to in the dark; this resulted in a 13% decrease in fruit set despite no change in visits to the plants by diurnal pollinators. Light pollution can also impact physiological variables in moths, as related to circadian rhythms and navigation (Bates et al., 2014). Furthermore, given that the reflectivity of clouds can amplify light pollution for several kilometers outside of city limits, Bates et al. (2014) speculate that moths may be drawn widely from somewhat distant rural locations into sub-optimal urban areas, thus potentially negatively impacting moth population abundance. Thus, the consequences affecting pollinators as a result of urbanization are multifaceted.

Selective horticulture involves the removal of perceived “pest” pollinator and plant species, primarily using insecticides and/or herbicides, as well as introduction of nonnative plants, which have the potential to become invasive. The effects of pesticides on pollinators aren’t fully parsed out, but there is evidence to support that various pollinator species are affected differentially by pesticide use (Barratt et al., 2015; Brittain et al., 2010). Brittain et al. (2010) found that species richness of butterflies and bumblebees decreased around areas that were more intensively farmed using higher pesticide loads. Native bees in particular are thought to be more sensitive to insecticide use (Brittain et al, 2010), potentially by exacerbating pathogenesis and compromising immune function in individual bees, which could then wipe out entire colonies (Potts et al., 2010). Even without directly selecting against native plants through pesticide uses, gardens riddled with nonnative species of plants could indirectly select against native plants. Alien plant species may outcompete native plants for pollinators, thus decreasing the efficacy of native plant reproduction (Morales & Traveset, 2009). Furthermore, nonnative

plants may negatively impact the survival and reproduction (e.g., oviposition, larval nutrition, larval growth, and larval survivability) of nearby pollinator species, thus leading to population decline (Sunny et al., 2015). Additionally, one must consider the potential for invasive species introduction; invasive plants substantially impact native biodiversity of plants, and as such, seriously affect the natural pollinators of the area (Cameron et al., 2012). Therefore, it is quite possible that while urban gardens may be intended to serve as oases within the concrete jungle of inner cities, they actually serve as death traps for many pollinator species.

While negative aspects of urbanization are rampant, some come with a silver lining. Although urbanized landscapes may act as sieves for reducing bee species, they may also function in promoting colonization of some bee species along an urbanization gradient; potential mechanisms for this include different availability of food and habitat resources in urban landscapes, as well as the presence of gardens with higher floral abundance and diversity than in more naturalized environments (Banaszak-Cibicka & Zmihorski, 2012). As an example of this, one study found that the diversity of the bee species within a city in Poland does not vary across an urban gradient (Banaszak-Cibicka & Zmihorski, 2012). Therefore, some researchers have turned to the exploration of urban wildflower gardens and green spaces to determine whether they may serve protective roles for pollinators within urban environments.

Urban Native Plant Gardens as Conservation Spaces

There is much debate surrounding the issue of whether native wildflower gardens within an urbanized landscape could be used as conservation spaces for pollinator populations.

Ultimately, it seems that the success of urban gardens in conservation efforts may depend on which set of pollinators is under study. For example, urbanization influences moth species abundance differentially: those species deemed the most vulnerable are negatively impacted to a

greater extent than the more resilient species (Bates et al., 2014). Similarly, as urbanization rises, butterfly species richness falls, especially in susceptible and more specialized species; as a result, such populations may drop 2.9-4.5% faster than the more generalist butterfly species populations (Clark, Reed, & Chew, 2007). However, another study that compared abundance of hoverflies to native bees, found that while hoverfly populations struggle in urban environments, native bee species generally thrive (Baldock et al., 2015). Therefore, it appears that urban wildflower gardens, while imperfect solutions, may serve to safeguard some pollinator taxa.

A garden's ability to provide a safety net for pollinator species is likely dependent on the characteristics of its surroundings within a city, such as the presence or absence of nearby urban greenspaces or other nearby gardens. The amount of vegetation near urban gardens, both as part of urban greenspaces and residential lands, is positively and directly correlated with the frequency of pollinator visits to the gardens, as well as the pollinator richness present (Matteson, Grace, & Minor, 2013; Hennig & Ghazoul, 2012). The maintenance of these greenspaces may also affect pollination networks. For example, Garbuzov, Fensome, and Ratnieks (2015) conducted a study wherein the frequency of mowing at a public park was reduced, which was associated with increased abundance of both flowers and flower-seeking pollinators. The introduction of strategically maintained greenspace may have a protective effect on the robustness of pollinator populations.

Whether urban gardens can act as a refuge for different pollinators may also depend on the characteristics of the individual gardens themselves, such as floral abundance and diversity, native plant diversity, and inclusion of specific host plants and nectar plants crucial for pollinator survival. Hülsmann et al. (2015) found that floral presence and the presence of certain plant species correspond with bumble bee species abundance and richness in urban environments

better than amount of impervious surface, i.e., the local factors of the garden outweighed the more global factors of the surrounding urban landscape. Floral resources have positive value for pollination networks, as floral abundance and diversity are positively correlated with number of pollinator visits to plants, thereby increasing the plants' reproductive success (Salisbury et al., 2015; Matteson, Grace, & Minor, 2013; Hausmann, Petermann, & Rolff, 2016; Matteson & Langellotto, 2010). Plant diversity within gardens is also positively associated with increased pollinator visits (Hennig & Ghazoul, 2012). Plant diversity can also be scrutinized through the lens of native and nonnative plant comparison. One study found that native plant species recruit a greater abundance of pollinators than nonnative species (Salisbury et al., 2015). However, nonnative plant species still function in the maintenance of pollinator richness (Matteson & Langellotto, 2011). Consequently, it appears that simply increasing the diversity of plant species within gardens should have positive impacts on pollinators. Finally, as a means to mitigate the effects of land use change on pollinator species through the cultivation of urban gardens, the most important aspects of gardens are the inclusion of both host and nectar plants (van Dyck et al., 2009); by providing nursery space for the eggs to hatch as well as food sources for both larval and adult forms, gardens with both host plants and nectar-producing plants assist in the restoration of the population size of threatened pollinator species, such as the monarch butterfly (Cutting & Tallamy, 2015). Changes in the structure of a garden as well as a garden's oversight have momentous influence on the services that it can provide, irrespective of the surrounding spatial context (Cameron et al., 2012; Shwartz et al., 2013).

Survivability in Urban Native Plant Gardens

Despite the fact that urban gardens have been explored in terms of the representation of pollinator species present within them, we currently know little about the survival of pollinators

in urban gardens versus rural habitats. Pollinators trapped in the concrete jungles of urban landscapes may seek out urban garden oases – but are these areas actually sanctuaries, or are they traps? Although research shows that the presence of basic needs such as food and shelter in urban gardens can assist in restoring the population size of pollinator species, (van Dyck et al., 2009; Cutting & Tallamy, 2015), we do not know how predation rates differ across these habitats.

The focus of our research is to determine whether urban gardens are more or less survivable for pollinator species as compared to natural ecosystems by examining predation rates on pollinator larvae. We created caterpillar models and placed them on native plants in gardens along an urban-suburban gradient. Native plants were chosen specifically to mitigate problems associated with host specificity of the native pollinator species. We used soft clay caterpillar models and assessed the models for damage from predators; previous studies have demonstrated that clay models resemble caterpillars enough to incite attacks from predators, and thus that they can serve as a means of estimating predation rates on caterpillars (Barber, 2012; Bateman et al., 2016). We hypothesize that urbanization factors into the survivability of pollinator larvae, thus impacting pollinator populations. Specifically, we predict that the models in urban gardens will exhibit greater damage than those in more suburban gardens. Our hypothesis is based on the general drop in nearby green resources that accompany increased surrounding impervious surface in urbanization: the urban gardens that are essentially isolated patches of greenspace offer less overall shelter, territory, and area to search for pollinator larvae as compared to suburban gardens, which have accessible nearby vegetation. As a result, pollinator predators in more urban areas are somewhat anchored to the islands of greenery in urban gardens for their own habitat space. Lack of nearby vegetation in urban areas would suggest that predators have a

smaller area to hunt for pollinator larvae, and therefore are likely more successful in combing through plants in the search for prey.

Methods

Garden Sites

We selected 24 garden and meadow sites, ranging in location from the center of Louisville to the rural outskirts of Jefferson County (see Figure 1). Study sites include both public and private native plant gardens, as well as public meadows. All sites contained primarily native plant species. We surveyed each garden to determine the plant species richness and plant biovolume density. The biovolume density was calculated in one of two ways. If the plants at gardens were far apart, the volume of each plant was taken individually and then summed. If the plants were tightly packed together, the heights of the plants were measured along transects two meters apart, and these values were averaged. The area of ground that the plants occupied was then divided out to get the biovolume density in m^3/m^2 . Additionally, to assess the “urbanness” of the study sites, we used GIS to calculate the percentage of impervious surface area in a circle centered on each of the gardens with a radius of 1.0 km. Gardens with greater than 40% surrounding impervious surface were categorized as urban, and gardens with 10-39% surrounding impervious surface were categorized as suburban. Garden characteristics are given for each garden site in Table 1; these include the percentage of impervious surface within a 1-km radius, garden area, plant biovolume density, plant species richness and garden type.

Caterpillar Models

To assess the predation rates on pollinators, we constructed caterpillar models out of green plasticine clay. Previous studies have shown that predation on models is acceptable

method to evaluate predation (Low et al., 2014). A nonhardening and nondrying material, this clay allows for impressions to be taken when predators attack the models. We used two sizes of caterpillars, one five centimeters long and the other three centimeters long, to assess whether predation differs with different instars of the caterpillars. The diameter of these models were three and five millimeters, respectively. We utilized an extruder to standardize the diameter of our models and then shaped the clay to mimic caterpillars in motion.

Forty caterpillars, twenty of each size, were deployed at each of the garden sites on focal plants. All focal plants in a garden were at least 1 m apart. In selecting the focal plants for the study, we chose only native plants; we attempted to use as many different native plant species as possible in each garden to reflect the plant species richness of the gardens. Each focal plant received one large and one small caterpillar model. These models were spaced forty centimeters apart in predetermined, randomized heights on the plants (top third, middle third, or bottom third of the plant). We secured them to the leaves of the plants using Loctite[®] super glue, and we revisited the sites to check for damaged caterpillars after 24 hours and 48 hours. All damaged caterpillars were removed after 24 hours, and all caterpillars were removed after 48 hours. The damaged models were categorized according to predator type, i.e. vertebrate (e.g., mammal, lizard, or bird), wasp, and spider; to determine predator type, we compared marks to those found on caterpillar models by Low et al. (2014). Examples of damage to caterpillars by different taxa are shown in Figure 2. A very small number of models were found to be damaged by ants and beetles; these were not considered further for analysis.

Statistical Analysis

All statistical analyses were performed in R. We tested to determine whether there were seasonal differences in the percentage of caterpillars damaged by different predator taxa

(vertebrates, wasps, and spiders) in July versus October, as well as whether the size of the models affected the proportion of damage observed on the models using chi-square tests. Because neither the seasonal differences nor the caterpillar size effects were significantly different ($\chi^2 = 5.40$; $p > 0.05$), we combined data from both seasons as well as from both model sizes for further analyses.

We used linear regression with GLM in R to analyze the effect of impervious surface on the proportion of model caterpillar damaged by each predator taxon. Due to collinearity, models could not include both impervious surface and garden area. We used a model comparison approach to determine which of these two variables better explained differences in the proportions of caterpillars damaged. The model with impervious surface had a lower AIC (-141.31) than the model with garden area (-136.16) and we accordingly used impervious surface as the explanatory variable in our linear regression.

Results

Overall, 23.55% of caterpillar models were damaged by potential predators, though the total amount of caterpillar damage did not vary significantly according to the percentage of impervious surface surrounding the models. Linear regression showed that plant biovolume and species richness did not have significant effects on the proportion of caterpillars damaged ($p = 0.96$ and $p = 0.77$, respectively). However, there was a significant interaction between impervious surface and taxon ($p < 0.0001$), i.e. the proportion of caterpillars damaged by each predator taxon was affected differently as the percentage of impervious surface changed (Figure 3). More specifically, as the proportion of impervious surface increased, the proportion of caterpillar models damaged by wasps increased, but the proportion of models damaged by

vertebrates and spiders declined. Parasitoid and predatory wasps were consistently responsible for a higher proportion of caterpillar damage than vertebrate or spider predators, regardless of the percentage of impervious surface surrounding the garden site.

Discussion

Damage rates on clay caterpillar models by different predator types were affected differently by urbanization, quantified as percentage of impervious surface. The proportion of damage by wasps increased in urban areas, but the proportion of caterpillars damaged by vertebrates and spiders decreased.

Urbanization and Vertebrate Species

The drop in vertebrate predation on the caterpillar models with increasing impervious surface may be explained in the general scarcity of mammals and birds in urban environments. Mammal species richness and relative abundance decline with increasing urbanization (Gortat et al., 2014; Łopucki & Kitowski, 2017; Gomes, Ribeiro, & Carretero, 2011). Even in small cities increasing impervious surface is correlated with drops in mammal abundance and richness (Łopucki & Kitowski, 2017). Potentially, this could be a result of worse living conditions for mammals in highly urban areas as compared to rural areas (Gortat et al., 2014). The type of urban greenspace also influences mammals' capacity to colonize and persist within the urban landscape (Gallo et al., 2017). Similarly, bird species richness and relative abundance decrease as impervious surface rises, and native bird species are especially affected by this trend (Silva et al., 2015; Villegas & Garitano-Zavala, 2010; Gagné et al., 2016). This trend may be due to the lack of resources available in urban landscapes (Melles, Glenn, & Martin, 2003) or perhaps increased air pollutant emissions in urban environments (Gagné et al., 2016). Moreover, with

increasing urbanization comes a shift in species composition from native to nonnative varieties, as well as diminished diversity (Blair, 1996). Therefore, the reduction in both number and heterogeneity of vertebrate predators in urban environments may account for decreased predation rates in urban gardens.

The fact that vertebrate damage on the models is higher in suburban settings is also supported by the literature. One study found that intermediate levels of urbanization, as displayed in suburban environments, have the highest levels of mammalian species richness, as compared to urban and rural areas (Riem et al., 2012). Bird diversity and species richness likewise peak in suburban settings (Blair, 1996; Chapman & Reich, 2007). The findings that vertebrate predators are most prevalent in suburban settings are consistent with the increased vertebrate damage on the caterpillar models.

Urbanization and Spider Species Richness and Abundance

Although links between vertebrate species and urbanization have been thoroughly researched, the effects of urbanization on spiders have not been well-studied. At least some components of the urban landscape (e.g., types of urban greenspace, urban warming, prevalence of arthropod prey) generally do not affect spider species richness or abundance (Moorhead & Philpott, 2013; Meineke et al., 2017). Evidence is mixed for how spider abundance changes on an urbanization gradient. Spider abundance and species richness was highest in disturbed urban habitats like gardens compared to more naturalized settings in urban areas (Philpott et al., 2014). However, ground surface spiders exhibit greater abundance in rural and suburban habitats than in urban habitats, potentially explained by amount of prey in each location (Fraser & Frankie, 1986). Additionally, the size of naturalized areas may have an effect, as spider diversity and abundance decrease with larger habitat fragments in urban areas (Bolger et al., 2000; Rebek,

Sadof, & Hanks, 2005). Clearly more research needs to be done on the effects that urbanization has on spider populations, given the paucity of information available.

In fact, spider species richness seems to be more dependent on local factors of gardens, such as floral resources, rather than landscape factors relating to the urbanness of the surrounding area (Otoshi, Bichier, & Philpott, 2015). Plant diversity in both urban and suburban habitats affects the community composition of aerial spiders (Fraser & Frankie, 1986). The research literature must be expanded to determine if such findings can be generalized to other spider species.

Urbanization and Wasp Species

When it comes to wasps and urbanization, there seems to be some mixed results, as the effects of urbanization may depend on which species are being examined (Morales et al., 2018). As shown with other insects (Bates et al., 2014; Clark, Reed, & Chew, 2007), specialized parasitoid wasps are more vulnerable to high land use intensity while generalist species are more resilient to such factors (Holzschuh, Steffan-Dewenter, & Tschamntke, 2010). One study found that parasitoid wasp diversity is negatively correlated with increasing impervious surface, decreasing as much as 10% in urban areas as compared to rural areas (Bennett & Gratton, 2012). However, another study found no distinction in wasp abundance or richness in relation to urbanization, as the families of various wasp species had been documented at all of their study sites along an urban-rural gradient (Christie & Hochuli, 2009). Yet another study, by Sumoski et al. (2009), reported that while wasp diversity doesn't change with urbanization, the abundance of parasitoids present in collected host species was significantly higher in urban sites than suburban or rural sites. Similarly, one study found increased wasp predation in areas with more urban land

cover (Morales et al., 2018). Given the contradictory information, more research must be conducted to determine how urbanization affects wasps.

Parasitoid wasp species abundance and richness have been shown to relate to local factors of gardens. Some of the most prominent local factors that are positively correlated with wasp presence include floral abundance and diversity as well as host abundance; these effects are especially pronounced in areas with moderate or high levels of urban development (Bennett & Gratton, 2012; Burks & Philpott, 2017; Rebek, Sadof, & Hanks, 2005; Morales et al., 2018). Additionally, Burks and Philpott found that garden size is positively correlated with wasp abundance while the size and diversity of plant species correlate with species richness (2017). Moreover, loss of plant diversity as well as a drop in the proportion of native plants is associated with falls in both abundance and richness of wasps (Raupp, Shrewsbury, & Herms, 2010); increased complexity of urban gardens through the introduction of more diverse and numerous plant species is associated with increased food availability and shelter for wasp species (Raupp, Shrewsbury, & Herms, 2010). The local factors of gardens could potentially offer some protective effects against urbanization.

The increased abundance of other predator types (e.g., vertebrate and potentially spider) in suburban lands as compared to urban lands could mean increased competition for larval prey for parasitoid wasp species. As such, the populations of wasps in suburban areas may be smaller than those in urban areas with less competition, which would result in a decreased proportion of damaged models in the suburban environment as compared to the urban environment. An alternate explanation could be that the predators of parasitoid wasps are negatively affected by urbanization; a lack of predation would mean a greater abundance of wasps exist in urban environments, which would thus correspond with increased wasp predation on the caterpillar

models. To fully parse out the dynamics affecting wasps in urban and suburban environments, more research needs to be conducted.

Urban Gardens as Conservation Space

Our results indicated that urban gardens, while different in predator type, did not differ significantly from suburban gardens in terms of the proportion of damaged caterpillar models. Therefore, urban gardens may function in pollinator conservation, at least for common pollinator species. Additionally, garden size did not significantly affect predation rates, meaning that gardens of any size may assist with urban biodiversity conservation. To maximize these potential resources, gardens should be well-managed. High floral diversity and abundance, as well as the complexity of the flora within are all important local factors of gardens associated with both vertebrate and invertebrate diversity (Goddard, Dougill, & Benton, 2010; Hülsmann et al., 2015; Salisbury et al., 2015; Matteson, Grace, & Minor, 2013; Hausmann, Petermann, & Rolff, 2016; Matteson & Langellotto, 2010; Hennig & Ghazoul, 2012; Otoshi, Bichier, & Philpott, 2015; Bennett & Gratton, 2012; Burks & Philpott, 2017; Rebek, Sadof, & Hanks, 2005; Morales et al., 2018). With proper care, urban gardens may aid in minimizing the population declines and extinctions of several pollinator species (Goddard, Dougill, & Benton, 2010). Future expansions of this study could include analysis of adult butterfly survival, as well as other pollinator taxa survival in urban landscapes.

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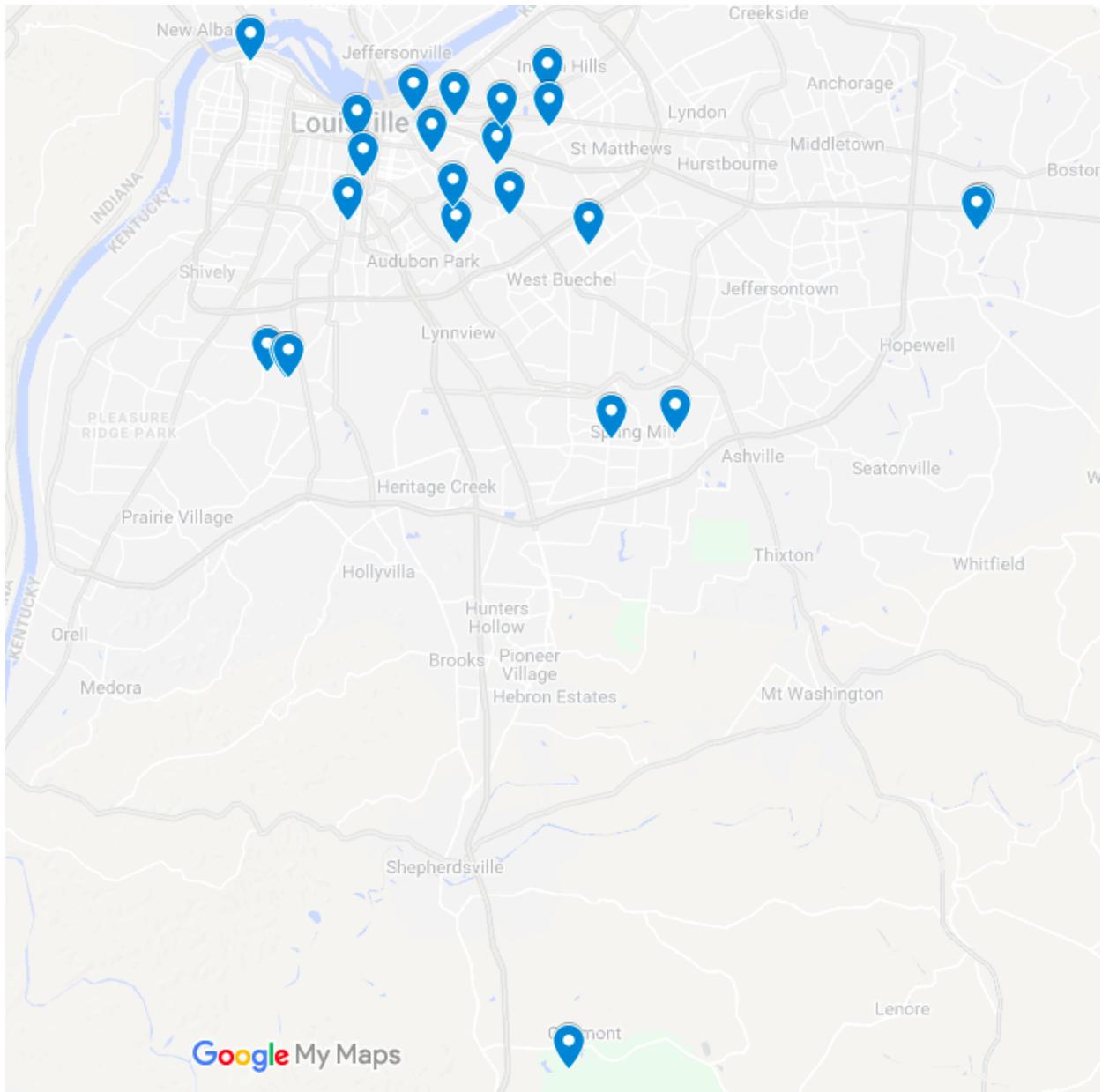


Figure 1. Map of Louisville, Kentucky with each of our 24 study sites marked (scale: 1 cm = 3.8 km).

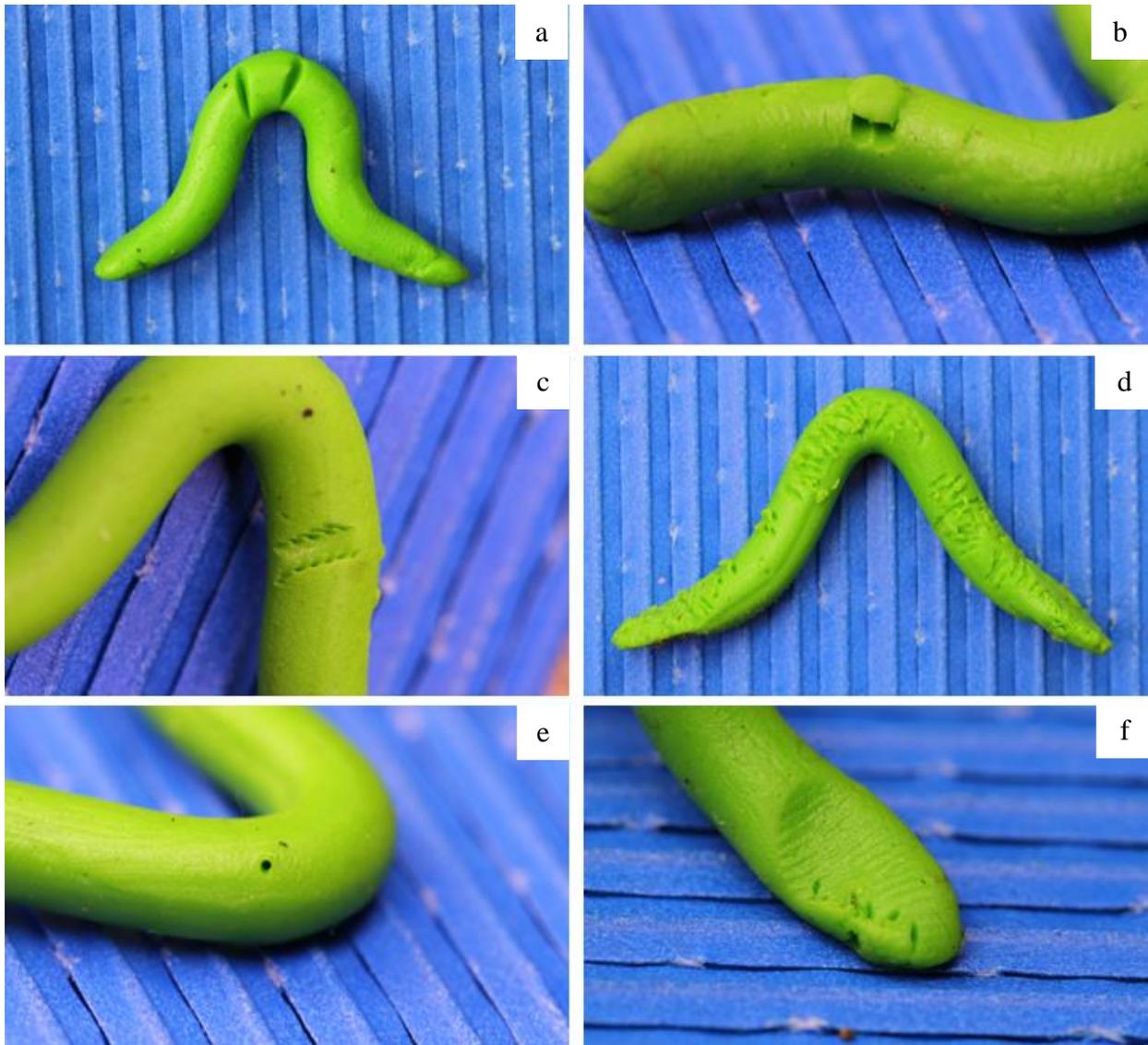


Figure 2. Examples of damage by predator taxa: a) bird; b) mammal; c) lizard; d) predatory wasp; e) parasitoid wasp; f) spider.

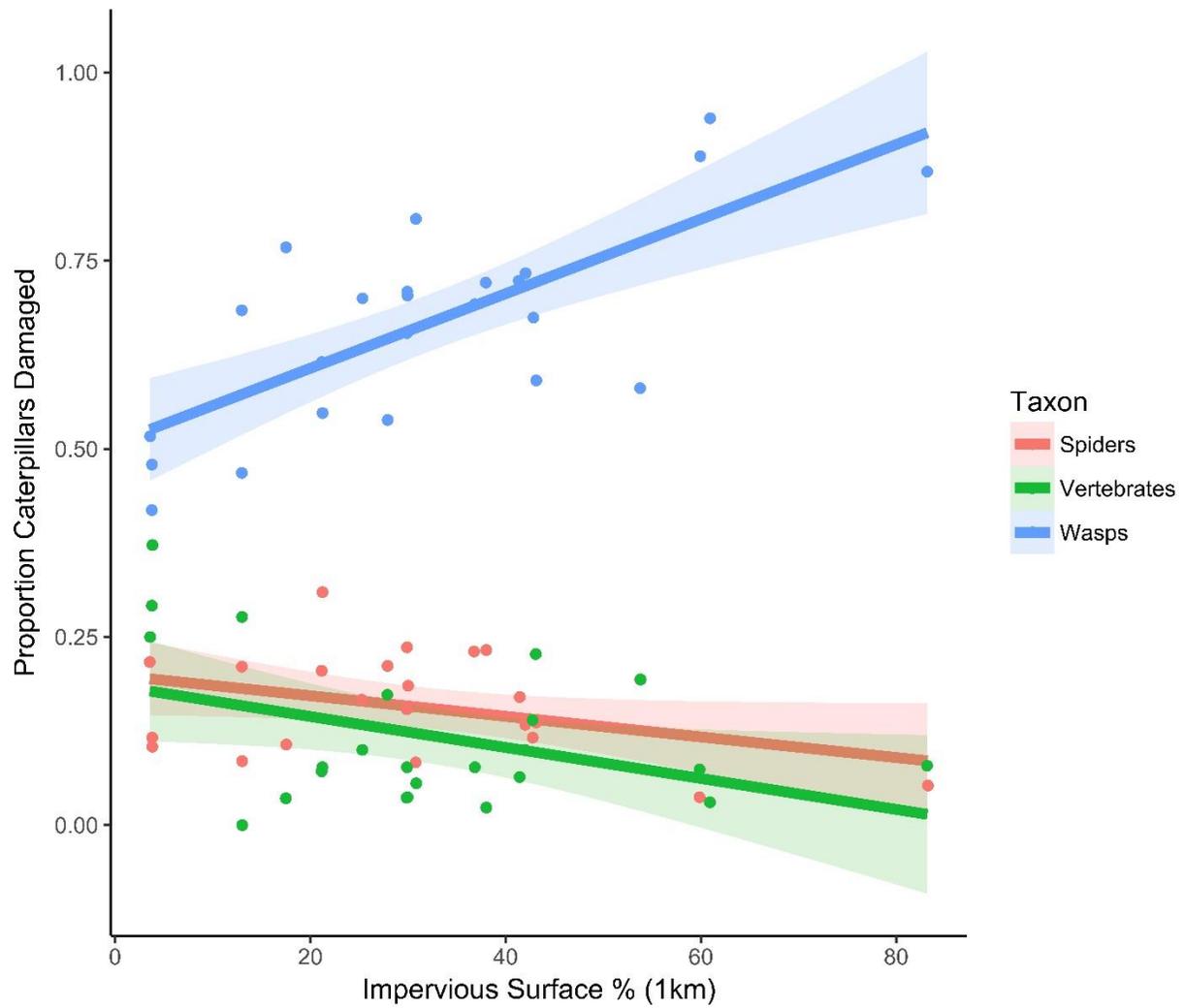


Figure 3. The proportion of spider and vertebrate predation on the caterpillar models decreased as the percent impervious surface increased, but the proportion of wasp predation on the models increased as the percent impervious surface decreased ($p < 0.0001$).

Site Name	Impervious Surface (1 km)	Garden Area (m ²)	BVD (m ³ /m ²)	Plant Species Richness	Type of Garden
Beckley I	3.8	89030.8	0.438	31	Public meadow
Beckley II	3.8	89030.8	0.500	29	Public meadow
Bernheim	3.5	129499.0	0.413	21	Public meadow
Carreiro	42.8	71.6	0.411	59	Private garden
Chenoweth	29.9	125.8	0.029	39	Private garden
Cherokee I	21.2	14164.0	0.381	24	Public meadow
Cherokee II	21.2	14164.0	0.403	28	Public meadow
Copper & Kings	53.8	341.9	0.406	38	Public garden
Word	36.8	96.8	0.299	49	Private garden
Eason	43.1	102.1	0.482	46	Private garden
Fuselier	17.5	20.9	0.403	24	Private garden
Holy Spirit	38.0	59.5	0.037	20	School garden
Iroquois I	13.0	97933.9	0.490	21	Public meadow
Iroquois II	13.0	97933.0	0.606	29	Public meadow
JPII	41.4	82.3	0.652	24	School garden
LNC	27.9	107.7	0.692	21	Public garden
Moore	25.3	384.1	0.546	22	School garden
OLCG	59.9	34.2	0.473	29	Public garden
Portland	30.0	39.1	0.037	33	School garden
Scroggins	42.0	35.4	0.127	40	Private garden
St. Agnes	30.8	82.3	0.045	26	School garden
St. Francis	83.2	184.5	0.242	13	School garden
Brown	29.9	453.0	*	49	Private garden
UofL	60.9	372.4	0.02094044	65	School garden

Table 1. Data collected from each of our garden sites, including percent surrounding impervious surface, garden area, biovolume density (BVD), and plant species richness. BVD was not available for the Brown garden because the owners did not want the disruption of a team of people walking through their garden.

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