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Author

Arnold, Joshua Earl

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By

Joshua Earl Arnold

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Professor Nick Mills
Professor Timothy Bowles

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Abstract

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Urbanized areas are the fastest-growing habitat worldwide. In the United States, over 80% of the population now lives in cities. It is expected that urban populations will continue to increase significantly in the coming decades. Matching urban population growth is an increase in urban food production; urban agriculture (UA) has grown 30% in the United States in the last three decades. Growing food in the city has become an increasingly common pathway to affordable, nutrient-rich, and culturally appropriate foods for people who live in high-cost cities.

A myriad of issues complicates urban agricultural production. Once urban farmers gain access to land, they must address many abiotic factors unique to cities that disrupt the ecosystem services many agroecological practices rely on. Increased impervious surface and decreased canopy cover affect hydrological and biogeochemical cycles and increase urban temperatures. Past land uses can affect soil quality and composition. These abiotic factors often exacerbate crop damage from herbivorous insects. Herbivorous pests in urban agriculture can become more persistent and increase in abundance in response to favorable environmental conditions in cities and cause damage to crops.

In urban agriculture, management of pests is almost universally accomplished through cultural practices as pesticides are rejected for environmental and health reasons. Many urban agriculturalists turn to agroecological pest management (APM) practices to increase on-farm beneficial insects and regulate pest populations. Agroecological practices proven on rural farms, such as crop diversification and floral resource provisioning, have been implemented to varying degree in the built environment, often with conflicting results. This research focuses on understanding urbanization impacts on agroecological pest management in urban agriculture. Specifically, how on-farm diversification schemes affect biological control services from parasitic Hymenoptera (PH). Recognizing how these biological control services function in fragmented urban landscapes is vital to urban farmers.

Understanding agroecological pest management practices and factors that may affect ecosystem function on urban farms necessitates understanding urban farm biophysical composition. Over three years, biophysical data were collected on twenty-nine urban farms in the San Francisco Bay Area. The physical composition of urban farms were measured, including overall size, areas of production, and percentage of land not in agricultural production. Indicators of specific management practices, such as type and percentage of mulch and ground cover, floral diversity, and crop and non-crop biodiversity, were recorded, and overall production was assessed. We found that practices associated with APM are widely adopted and are often practiced concurrently. Our research shows that urban farms are highly productive, and most crops grown feed local community members. Land use and spatial composition of urban farms varied, but the production area as a percentage of the total area is often low, and areas set aside for pollination gardens or beneficial habitat are common.

As agroecological pest management in urban agriculture is an understudied topic, a systematic review of research specific to UA and biological control services was conducted. Previous findings recorded significant impacts on both natural enemy and herbivorous pest populations in response to landscape and local effects, but findings remain inconsistent. Local management factors related to agroecological practices, including increased floral abundance, mulch and leaf litter, high plant species richness, and structural diversity, had significant beneficial effects on natural enemy abundance, richness, and biological control services.

We conducted a two-year experiment testing the effects of local management practices and landscape effects on parasitic Hymenoptera, aphids, and crop damage on common *Brassica* crops. Two fundamental hypotheses in conservation biological control: the enemies and the floral nectar provisioning hypotheses, were tested in novel urban agroecosystems. Local and landscape factors were measured and assessed for their influence on PH populations on eleven San Francisco Bay Area urban farms. Farms were selected to represent a variety of sizes and surrounding imperviousness. Our research indicated that Local factors, including increased mulch coverage, crop richness, and percent of non-crop areas, were predictors of increased PH abundance and aphid parasitism rates.

To test the effects of floral provisioning on PH we sampled thirteen common floral species across community partner sites to link common floral species in urban farms to PH families and subfamilies known to utilize aphids as hosts. We found that PH had no feeding preference, and floral species had little impact on PH abundance. To assess the second criterion of the nectar provision hypothesis, a demonstrable reduction in pests or crop damage, we looked at aphid abundance, rates of parasitism, and overall crop damage on brassicas. Our results show that farms with increased floral richness have lower aphid counts per plant. Our findings indicate that on-farm habitat manipulations can increase ecosystem function, supporting the enemies hypothesis in fragmented urban agriculture sites.

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Introduction

Growing food in the city is not uncommon, but it has rarely been straightforward or without controversy. Historically, the practice has been implemented, supported, and championed during periods of domestic turbulence and economic stress. Conversely, urban agriculture has often been left with little support when "crises" have subsided, or even actively suppressed (Lawson, 2005). Post-WW2, urban agriculture (UA) has been especially prevalent in the context of social movements (Caruso et al., 2016). As worker wages stagnated in the late 60s and 70s, and access to culturally relevant foods became more difficult for rent-burdened people, urban agriculture became a means to supplement diets, especially in BIPOC communities. When the Black Panther Party (BPP) started community garden networks to support their Free for Children Breakfast Program in the late 60s, the Hoover administration said that the program was the "most influential activity going for the BPP and, as such, is potentially the greatest threat to efforts by authorities to neutralize the party." Fifty years later, Billy X Jennings, the official archivist of the BPP said that the Breakfast Program was "one of the biggest and baddest things [the BPP] ever did" ("'One of the Biggest, Baddest Things We Did,'" 2019).

Oakland, California, was the epicenter of urban agriculture in the 60s. Presently, in Oakland and many other San Francisco Bay Area cities, urban farmers continue to farm the city. However, urban farmers continue to face challenges. Despite the inclusion of urban agriculture and sustainable urban food production in many city plans, formal policy and financial support are often lacking (Horst et al., 2017; A. B. Siegner et al., 2019). Often, urban agriculture is relegated to degraded vacant lots on the margins of high-value urban land. Even though urban agriculture is very productive, it fails to generate substantial profit at scale, therefore, falls outside the concept of "highest and best use", creating an economic disincentive for support (Clinton et al., 2018; Drake & Lawson, 2014; Horst et al., 2017). A lack of secure tenure affects the implementation of long-term practices and investment in farming operations. Despite these difficulties, urban agriculture seemingly thrives – growing every decade and providing fresh and culturally appropriate foods to a wide range of diverse urban communities (Lawson & Drake, 2013; A. Siegner et al., 2018).

While urban agriculture exists in the margins of urban land use cycles, it also persists in a nexus of social, economic, and ecological factors that co-create and shape this land use. These social-ecological and biophysical factors can often seem insurmountable when creating and maintaining urban agroecosystems. Compacted soils must be remediated, and entire ecosystems created out of vacant lots, old sports fields, and former parking lots. Many innovative cultural technologies and practices are implemented during the creation and maintenance of urban agriculture sites. Every urban farmer must draw on a multitude of cultural technologies and agroecological knowledge to overcome the technical barriers of creating thriving agroecosystems that feed their community (Lin et al., 2015). One persistent issue encountered in urban farms is pest outbreaks and crop damage. Urban farmers often choose to manage pest outbreaks through cultural and mechanical practices that rely on regulating ecosystem services provided by naturally occurring "enemy" arthropods that predate or parasitize crop pests. During initial stages of the research project outlined in Chapter 1 of this dissertation, urban farmers demonstrated a great interest on the ecological management of crop pests, and questioned whether on-farm management practices, such as floral provisioning and agroecosystem diversification, would be impacted by the conditions in the built environment and, if so, whether on-farm management practices could essentially offset these impacts.

Concerns about urbanization effects and pest management are not unfounded. Several conditions exist in urban areas that can impact ecosystem function regarding pest management. Firstly, urbanization fragments the landscape and creates a selection pressure for disturbance tolerant species (Faeth et al., 2011). Frequently these disturbance tolerant species are herbivorous insects who impact urban cropping systems. Herbivorous insects in urban areas have been documented to be more fecund, larger, and persist for longer (Dale & Frank, 2014, 2018; Korányi et al., 2022; Parsons & Frank, 2019; Turrini et al., 2016). Secondly, natural enemies have been shown to suffer detrimental effects from urbanization, disallowing them to reduce or stabilize urban herbivore populations in urban agroecosystems (Burks & Philpott, 2017; M. Egerer et al., 2018; Lagucki et al., 2017; Morales et al., 2018; Philpott et al., 2014). Because urban farmers typically employ agroecological management practices for pest control, these effects are especially compelling.

The research reported in this dissertation focuses on developing a better understanding of urban agroecosystem composition in urban farms, especially in the context of factors that may be important for biological control services. Several questions were developed in collaborations with urban farmers during the first two years of this project and influence the entirety of this dissertation.

How do urban farms ecologically function?

Urban farms have been documented as highly biodiverse (Clucas et al., 2018; Goddard et al., 2013; Sperling & Lortie, 2010), despite existing within landscapes that typically reduce overall biodiversity (Faeth et al., 2011). On-farm management practices predominantly determine urban farm biodiversity. These management practices can influence crop biodiversity, structural complexity, and soil fertility and provide valuable habitat for flora and fauna in disturbed landscapes. As a result, management practices and on-farm composition can influence ecosystem function both on- and off-farm. The ecological composition of UA, the agroecological practices implemented on farms, and on-farm spatial composition have often been overlooked in urban agroecology studies (Lin et al., 2015). Chapter 1 reports on two years of research on this topic and provides context regarding on-farm elements including management practices and spatial composition on 29 farms in the San Francisco Bay Area, California.

Does urbanization perturb ecosystem function and impact biological control services on urban farms?

The landscape composition surrounding farms can be an important determinant for species persistence in these diverse habitats (Kruess & Tscharntke, 1994, 2000; Prugh et al., 2008; Tscharntke et al., 2012, 2012). Urban arthropods must derive enough resources within the landscape matrix surrounding these farms to traverse these chaotic landscapes and persist as metapopulations (Driscoll et al., 2013). As many arthropods provide essential ecosystem function, especially in the context of agroecological management practices, their occurrence, or lack thereof, can impact ecosystem function on urban farms. How urban landscapes affect biological control services on urban farms is still being explored. In Chapter 2 of this dissertation, we perform a systematic literature review of on- and off-farm impacts on natural enemy and pest abundance, richness, and ecosystem service provisioning.

Can farmers overcome impacts of urbanization through on-farm management practices?

If landscape factors are a determinant of on-farm ecosystem function, the question then broadens, and we must question whether urban farmers can overcome these impacts through on-farm management practices. Diversification practices have been well-documented in rural farms to lead to effective agroecological pest management, but these effects are less clear in urban agroecosystems (J. E. Arnold et al., 2019; Karp et al., 2018). In Chapters 3&4 we test two fundamental theories understudied in urban agroecosystems, the enemies hypothesis (Root, 1973) and the floral nectar provisioning hypothesis (Heimpel & Jervis, 2005), using parasitic Hymenoptera as a research organism. We look at on-farm management practices and off-farm spatial composition to better understand these local and landscape effects that affect parasitic Hymenoptera abundance, richness, and resulting ecosystem services with the goal of understanding how on- and off-farm factors affect biological control services.

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Chapter 1: On-farm Spatial Composition, Management Practices, and Estimated Productivity of Urban Farms in the San Francisco Bay Area

Abstract

Urban areas are the fastest growing land type worldwide. By 2060 it is expected that ~70% of the human population will live in cities. With increased urban population growth, food sovereignty and security issues have gained more attention, resulting in a drastic increase in urban food production activities including, urban farming and gardening. The extent to which urban farms function, their social, ecological, and economic composition, and their overall impact on local food security has become an often overlooked, but important topic. From 2014 to 2017, we partnered with twenty-nine urban farms in the San Francisco Bay Area for a broad-scale survey of urban farm characteristics. Findings reported in this research focused on local (on-farm) characteristics, including management practices, on-farm spatial composition, and estimated productivity. We implemented open-ended surveys for farm managers to better understand management practices, measured on-farm elements, including yields, crop biodiversity, weed composition, and abundance, and measured spatial characteristics such as area of production, non-crop area, and proportion of infrastructure to better understand how urban farms were spatially configured. We found trends regarding spatial composition, including a large proportion of farm area dedicated to infrastructure and underutilized potential production space. All farms surveyed had adopted a breadth of agroecological management practices, including cover cropping, crop rotations, intercropping, and a range of soil conservation practices. Measured farms are incredibly productive, with estimated seasonal yields of 7.14kg/square meter. Estimated yields were comparable with actual yields as measured at two participating farms.

1. Introduction

Urban agriculture (UA) sites, herein identified as urban farms, are co-created by the immeasurable factors that occur in the built environment. Not only do ecological factors, like soils, and climate impact their function, but social and economic processes shape their location, size, and even what they produce (Caruso et al., 2016; Lawson, 2005). These farms occur in vacant lots, schools, city parks, and other underused urban spaces; and span a variety of typologies, including institutional urban farms, small allotment style gardens, collectively managed spaces, and many distinct combinations in between (Lin et al., 2015; McClintock, 2010). Each urban farm has a network of social and economic support structures situated in a unique ecosystem that, in sum, influence form and function (Lin et al., 2015; Mougeot, 1999; Zezza & Tasciotti, 2010). Despite UA's prevalence in the modern urban landscape, little is known about on-farm composition, management, and function; or whether trends exist across the different categories.

The San Francisco Bay Area is a historical focal point for urban agriculture initiatives, often affiliated with food sovereignty and food justice organizations that currently operate urban farms throughout the East Bay. To better understand the form and function of urban agriculture systems, we initiated the Urban Agroecology survey in 2014 and engaged with urban farms and their managers in the San Francisco Bay area for three years. Our research goal was to better describe urban agriculture's local composition and practices by investigating the on-farm characteristics of urban farms, including landscape composition, prevalence of management practices, and a variety of production measures, including estimated yields, biodiversity, and disposition of harvests.

1.1. On-farm composition

Investigations regarding local and landscape spatial features of urban farms have been mainly constrained to three foci: the overall size, surrounding landscape, and ecological composition of the farm, especially in the context of species abundance or diversity and species-mediated biodiversity and ecosystem services (B&ES) (J. E. Arnold et al., 2019). Urban farms are often framed as potential habitats (i.e., habitat patches) in urban areas and are studied from the existing literature on species-area relationships in fragmented landscapes (Tscharntke et al., 2012). Generally speaking, species diversity is reduced in urban and peri-urban landscapes, and the abundance of urban-tolerant species increases (Faeth et al., 2011; Gaertner et al., 2017; Kruess & Tscharntke, 1994). These dynamics are incredibly complex and confound research regarding species occurrence, persistence, and species moderated ecosystem services in urban environments (Tscharntke et al., 2005, 2012). Despite conflicting results regarding species-area relationships and B&ES in fragmented landscapes, the size of urban farms has been a significant explanatory variable for a variety of measured ecological phenomena. Farm size has been found to influence ecological function, especially provisioning of regulatory ecosystem services like biological control services and mediation of abundance and diversity of natural enemy and herbivorous pest species (Burks & Philpott, 2017; Christie & Hochuli, 2009; M. H. Egerer et al., 2017; Matteson & Langellotto, 2011; Morales et al., 2018; Otoshi et al., 2015; Philpott et al., 2014; R. M. Smith et al., 2006; Sperling & Lortie, 2010).

Landscape analysis is also common and similarly framed from a B&ES approach to and from urban farms with urban arthropods and resulting regulatory ecosystem services. These studies typically measure surrounding imperviousness at varying distances and resulting impacts to natural enemy abundance and diversity (Burks & Philpott, 2017; M. Egerer et al., 2018; M. H. Egerer et al., 2017, 2017; M. H. Egerer, Liere, Lin, et al., 2018; Mace-Hill, 2015; Morales et al., 2018; Otoshi et al., 2015), herbivorous pest abundance and diversity (M. H. Egerer, Liere, Lin, et al., 2018; Lagucki et al., 2017; Lowenstein et al., 2016; Lowenstein & Minor, 2018; R. M. Smith et al., 2006), or focus specifically on regulatory ecosystem services (Philpott & Bichier, 2017). Despite the obvious utility of these studies in understanding ecological function on urban farms, regulatory ecosystem services, and overall urban landscape ecology, the available landscape data are often criticized for their low resolution. The available technology cannot discern the fine detail of off- or on-farm composition (Qian et al., 2015), necessitating an on-the-ground approach to complement further research and better understand on-farm composition.

Previous studies that have measured landscape effects to urban agriculture ecosystem function have also incorporated local, on-farm, structural factors such as occurrence, the height of, and canopy cover of perennials and overall structural diversity (J. E. Arnold et al., 2019; Burks & Philpott, 2017; M. H. Egerer et al., 2017; Lagucki et al., 2017; Lowenstein et al., 2016; Morales et al., 2018; Otoshi et al., 2015; R. M. Smith et al., 2006). Incorporation of these structural factors and other features commonly associated with management practices such as floral provisioning into their analysis have shown significant effects on species occurrence, but these factors are often measured in isolation of other spatial land uses in UA.

While landscape and local factors demonstrably have an impact on ecosystem function, to the authors' knowledge, no studies expressly incorporate the "patches within patches" concept in UA and include on-farm land-use composition (i.e., the area of production, infrastructure, unused spaces, or areas set aside for natural habitat) in their analysis. In short, existing literature that accounts for both on- and off-farm effects on ecosystem function treats on-farm spatial composition as homogenous. Failure to account for spatial composition in landscape ecology studies can have significant implications for the validity of explanatory factors in these investigations as density-dependent, and species-area relationships should depend on the farm's spatial composition. Furthermore, the size and production capacity are essential factors for the planning and design of urban farms, as urban agriculture is a persistent and growing land use in urban spaces that directly impacts urban food security (Horst et al., 2017; A. Siegner et al., 2018). How these patches of agricultural production are managed in urban areas can have significant consequences regarding the future of UA and ecological studies within these systems. For this research, we investigate the preliminary aspects of these future research questions, seek to determine the overall spatial composition of urban farm land-use, and investigate whether specific trends regarding land-use types exist in our participating farms.

1.2. On-farm management practices

Urban farms exist within a landscape that has been co-created through social, economic, and ecological processes. Not unique to but prevalent in urban farms are a variety of abiotic and biotic conditions that offer unique challenges for urban farmers. Often, farmers can mitigate or adapt to challenges encountered while urban farming by implementing specific agroecological practices. Our survey sought to understand better how urban farmers manage their farms from an agroecological perspective and the extent to which management practices have been adopted or utilized in these unique landscapes. Implementing specific diversification and management practices can directly impact soil quality and mediate regulatory B&ES essential for ecological management of soils, pests, and weeds.

Urban farmers are often practicing on ruderal or underused urban landscapes that often have a variety of factors that decrease urban soil biodiversity and function (Wortman & Lovell, 2013). Urban soils typically have a higher bulk density, lower amounts of organic matter, and higher levels of debris and contamination (Lehmann & Stahr, 2007). Moreover, urban soils have lower levels of nutrient cycling due to disruptions or reductions of biological materials like leaf-litter and water cycling (White & McDonnell, 1988). Of note, areas where urban agriculture may be most impactful from a food security standpoint, are also areas with lower levels of tree canopy and greenspace (Casey et al., 2017; Heynen, 2006), important aspects of nutrient cycling in natural ecosystems. These factors often create poor conditions for agricultural production and require intensive management to restore soil quality and fertility.

Several agroecological practices have been shown to have significant impacts on soil composition and function in non-urban agroecosystems, including the application of organic matter (compost, manure, and other organic amendments) (Loper et al., 2010), crop rotations (Bowles et al., 2020), cover cropping, and no-till practices (Feng et al., 2021; Sharma et al., 2018). The extent to which these practices have been implemented in urban agriculture has been understudied. Still, urban farmers have widely adopted these practices under the auspices of restoring the ecological function of urban soils. In addition to these practices creating favorable conditions in agroecosystems, they also are primarily rooted in indigenous agricultural practices that rely on ecological management, eschewing chemical inputs such as synthetic fertilizers and pesticides. Reducing the need for off-farm inputs and focusing on ecological management practices are essential aspects of UA ethics that focus on ecological and individual health.

Many of these practices are beneficial to restoring the tilth to urban soils and have many beneficial effects on pest and weed management. Herbivorous pests in urban areas can be challenging to manage due to increased urban temperatures (Taha, 1997), stressed urban plants (Dale & Frank, 2014, 2018; Parsons & Frank, 2019; Turrini et al., 2016), the prevalence of disturbed areas often colonized by weedy alternative host species (Gaertner et al., 2017). In addition to being beneficial to soil health, diversification practices such as intercropping, crop rotations, and floral provisioning have been shown to have beneficial effects on natural enemy populations, reduce herbivorous insect abundance, and decrease crop damage (Landis et al., 2000; Letourneau et al., 2011). Moreover, soil management practices are inexorably linked to biological control of weed species, especially granivory, and disruption and competition of weed species (Liebman & Davis, 2000; Lundgren, 2005; Sarabi, 2019).

The extent to which agroecological management practices regarding soil health, pest, and weed management, have been adopted and practiced by urban farmers is unclear. Understanding adoption of and implementation of these practices can be of great importance to understanding how urban agroecosystems function and how urban farmers can overcome challenges unique to the built environment.

1.3. Productivity and crop biodiversity

Urban agriculture is often considered within the context of a production paradigm, value derived from urban farms is contextualized in a production mindset despite the myriad of measured benefits to the community who engage with these spaces (Horst et al., 2017; A. Siegner et al., 2018). While productivity is not explicitly a goal of most urban agroecosystems (J. Arnold & Rogé, 2018), it is an important aspect of food security (A. Siegner et al., 2018). That being said, production capacity is often cited when both justifying and criticizing UA's efficacy, and continues to be a salient topic in UA discussions (Clinton et al., 2018; Martellozzo et al., 2014). Many studies have attempted to address productivity, both through direct measurements of yields and yield estimates in an effort to better understand urban agroecosystem production capacity (Colasanti et al., 2010; Gittleman et al., 2012; McDougall et al., 2019; Wekerle & Classens, 2015).

Agricultural yields in urban agroecosystems have been notoriously difficult to measure. Because these spaces often operate outside of traditional crop planning, harvesting often occurs sporadically and outside the scope of top-down farm management, making it difficult to capture overall yield of each plant. Moreover, many crops grown in UA systems are harvested repeatedly throughout the growing season, so one engagement with the plant is unlikely to represent its true yield. Another difficulty often encountered in this process is that removing crops for measurement is a significant impact to the farm. Several strategies have been implemented in the previously cited studies, including estimates and direct counting and weighing of crops. While estimates are not a particularly effective way of understanding true production, they can be a useful tool for estimating potential yields. For the purpose of this research, we used the estimated productivity per plant to create a productivity measure per unit of area to gain a better understanding of overall potential yields in UA.

Related to productivity, disposition of harvest is an important factor connecting productivity to food security. Most often, research that has measured productivity on urban farms has failed to gain insight into where harvests go, and how they are utilized. Most literature regarding urban food production and its ultimate fate rely on theorized or hypothetical food distribution scenarios but fail to understand how and if urban farm production impacts food insecurity or alleviates food impoverishment (A. Siegner et al., 2018). In this research we work directly with farm managers through open-ended surveys to gain insight on the ultimate fate of urban farm production.

1.4. Research efforts

Despite these differences and typologies, and myriad of challenges and benefits derived from such systems, little is known about the prevalence of certain management practices, on-farm spatial composition, or trends regarding the productivity of these spaces. This agroecological survey attempts to clarify these aspects of urban farming to gain a better insight into urban agroecosystem function, composition, and productivity - this information will hopefully be of great utility to urban planners, urban farming advocates, and ecologists studying these important and complex managed ecosystems.

2. Methods

2.1. Study Design

Initial outreach to local urban farms in the East Bay area of San Francisco started in 2014. Potential community partners were identified through a review of local news, social media, and websites that identified urban farmers and community gardens that were impacting local food security. After initial contact an initial meeting was scheduled for researchers and farm managers. During the first visit, researchers administered an open-ended interview that included 54 questions based on management practices, social and economic factors, and ecological phenomena (see Appendix A1). Research questions qualify as exempted from Institutional Review Board approval as per the criteria on research set forth by the Committee for Protection of Human Subjects (CPHS) and Office for Protection of Human Subjects (OPHS) (*UC Berkeley Committee for Protection of Human Subjects*, n.d.). Additional community partners were identified during this process during research visits and referrals from initial contacts. Research partners and affiliated urban agriculture sites were asked to be community partners if they met two criteria: 1. Level of community engagement (i.e., operations were open to community participation and community members derived some benefit), and 2. They were not explicitly for-profit, production-centric operations. Some selection bias probably occurred during this time as organizations that had a full or part-time farm manager were more likely to sustain contact with researchers. Nineteen sites were identified for research in year one. Researchers visited several times throughout the summer and fall growing season as approved by farm managers.

In 2015 we saw some attrition from previous farms due to low response rates and difficulty in scheduling. Four additional research partner sites were included in 2015. All measures except open-ended surveys were repeated in 2015. Subsequently, in proceeding research years six additional sites were added for a total twenty-nine farms that participated in this research (Figure 1). Due to variability in support, turnover of garden managers, and a variety of other factors, some farms only participated in portions of the research (see Appendix A2). During the duration of the urban agroecology survey, six of our community partner sites were abandoned and/or developed.

Figure 1. Urban farm study sites (*N=29*).

2.2. On-farm composition

To determine on-farm composition of research sites, we physically measured urban farm size, area of production, non-crop areas, and areas used for infrastructure. Total farm size was measured using Google Earth Pro and ground-proofed during site visits. Farm production space was measured by hand and included all space in the gardens used for producing crops (both annual and perennial). Not all production occurred in-ground, therefore the overall estimate of area used for production included both raised garden beds and in-ground production. Non-crop areas are defined as managed areas not primarily used for food production and were often set aside as pollinator or natural enemy habitat. These spaces included a variety of perennials and annuals, flowers, and other non-crop features. Infrastructure was defined as area not being used for production, or conserved for non-crop habitat (includes buildings, pathways, etc.) and can generally be considered areas utilized for other on-farm uses. These measurements were translated into proportions of total farm area for analysis (Table 1).

Table 1. Farm size and Land use composition.

Farm Size and Land use Composition (n=19)							
Site #	Total size (m^2)	Production (m^2)	Non-crop (m^2)	infrastructure (m^2)	Production (%)	Non-crop (%)	Infrastructure (%)
1	95	10.6	12	72.4	0.11	0.13	0.76
2	117	31.3	31	54.7	0.27	0.26	0.47
3	140	24.1	24	91.9	0.17	0.17	0.66
4	175	58.36	30	86	0.33	0.17	0.49
5	394	255	12	127	0.65	0.03	0.32
6	522	296.72	100	125.28	0.57	0.19	0.24
7	537	25	25	487	0.05	0.05	0.91
8	566	42	0	524	0.07	0.00	0.93
9	664	300	49	315	0.45	0.07	0.47
10	728	136	119.75	472.25	0.19	0.16	0.65
11	778	96.5	106.7	574.8	0.12	0.14	0.74
12	964	184	0	780	0.19	0.00	0.81
13	1367	867	90	410	0.63	0.07	0.30
14	2348	760	188	1400	0.32	0.08	0.60
15	2443	511	200	1732	0.21	0.08	0.71
16	2603	405.18	440	1757.82	0.16	0.17	0.68
17	3105	973.5	800	1331.5	0.31	0.26	0.43
18	4477	966	595	2916	0.22	0.13	0.65
19	8016	7775	0	241	0.97	0.00	0.03
Average	1581	722	149	710	0.32	0.11	0.57

2.3. Productivity and distribution

Estimated Productivity (EP) was measured by randomized quadrat counts of land in agricultural production. All crop plants within the quadrat were identified to cultivar. Each quadrat was also identified for production practices, whether the bed was raised or in-ground, and type of irrigation (drip irrigation or hand watered). Estimated productivity was measured by the number of plants per quadrat. Per plant yields were limited by available data and a diversity of sources were used to estimate yields including How to Grow More Vegetables (Jeavons, 2012) and unpublished yield data from field trials at the Oxford Tract Research Station at UC Berkeley. The methodology used to estimate the average potential yields per square meter are based on prior work in urban agriculture (*The Potential for Urban Agriculture in New York City.*

Growing Capacity, Food Security, & Green Infrastructure, 2012; Colasanti et al., 2010; Gittleman et al., 2012; Vitiello et al., 2009).

Distribution of harvests were self-reported by farm managers during open-ended questionnaires. Farm managers were asked to report proportions of harvest that went home with people who worked on the farm or were donated to farm neighbors, crop sales through farmers markets, and donations to community food organizations such as food banks.

2.4. Crop diversity and weed occurrence

Crop diversity was measured using two methodologies, productivity quadrats and eight-meter transects, to account for in-bed diversity as well as on-farm diversity. Both methodologies counted all visually identifiable crop plants, and non-crop plants (flowers). Iterations of transect counts were determined by overall farm size with larger farms requiring six to nine transect counts and smaller farms only three. Crop biodiversity was measured five times on each farm over the duration of the research. Weed abundance and diversity were measured using in-bed quadrat counts and categorized as broadleaf and or grasses. More developed weeds were identified to morphospecies.

2.5. Management practices

Information on common land management and farming practices (see Appendix A3) such as crop rotations, cover cropping, use of mulch, intercropping, on-site composting, soil management practices, pest control strategies, weed management were collected during the open-ended survey (Table 2). Confirmation of practices implemented on the farm were ground-truthed over several visits to the farm. In some cases, community farms were managed individually rather than collectively. In most cases we observed common practices among plots and generalized these as commonly used on the site, however, not all participants can be expected to use uniform management practices, and not all practices are visually observable, especially in the context of soil amendments and pest management practices (excluding pesticide use which was always prohibited on all research sites). For analysis, practices including crop rotations, cover-crops, intercropping, mulching, application of soil amendments including compost, manure, and fish emulsion, composting on-site, and no-till practices were aggregated to create an overall management-intensity index.

Table 2. On-farm Management Practices.

On-farm Management Practices (n=29)						
Agricultural practices						
Inter-cropping	Cover-cropping	Mulch	Rotations	Green manure	Double-dig	No-till
27	25	19	18	7	6	1
93%	86%	65%	62%	24%	20%	3%
Soil management						
Compost	Manure	Fish emulsion	Compost tea	Worm-castings	Minerals	-
24	11	10	9	5	3	-
82%	38%	34%	31%	17%	10%	-
Animal Integration						
Bees	Chickens	Worms	Goats	Fish	Ducks	Rabbits
10	7	4	3	2	1	1
34%	24%	14%	10%	6%	3%	3%

2.6. Data analysis

Data analysis focused on the spatial composition of farms and how spatial composition, including proportions or production space, natural habitat, and infrastructure were related to the overall farm size. We analyzed on-farm spatial composition using regression and classification trees (CART) in R (Therneau et al., 2022). Management practice occurrence were measured and used as explanatory variables for proportion of weed coverage (weed density), and estimated productivity (EP). Weed density was checked for normality using a Shapiro-Wilks Test, and for Homogeneity of Variance using Levene's Test. Weed data were analyzed using a Non-parametric Kruskal-Wallis Tests with Dunn’s post hoc analysis.

3. Results

3.1. On-farm composition

For analysis we questioned how on-farm land use categories and overall size influenced the proportion of production, natural habitat, and infrastructure. We used classification and regression trees (CART) to look at overall predictors of the three land use categories. CART analysis indicated that the proportion of production was best predicted by overall proportion of on-farm infrastructure. Eighty-five percent of farms had over 40% of their overall area committed to farm infrastructure, and 58% of those farms had infrastructure in excess of 62% of overall farm size. Infrastructure was by far the largest on-farm land use category, accounting for an average of (57%) in all farms measured (Figure 2). When accounting for other non-production land use, an average of 68% of on-farm area was not utilized for food production. Overall size of farms was a poor predictor of any other land use type.

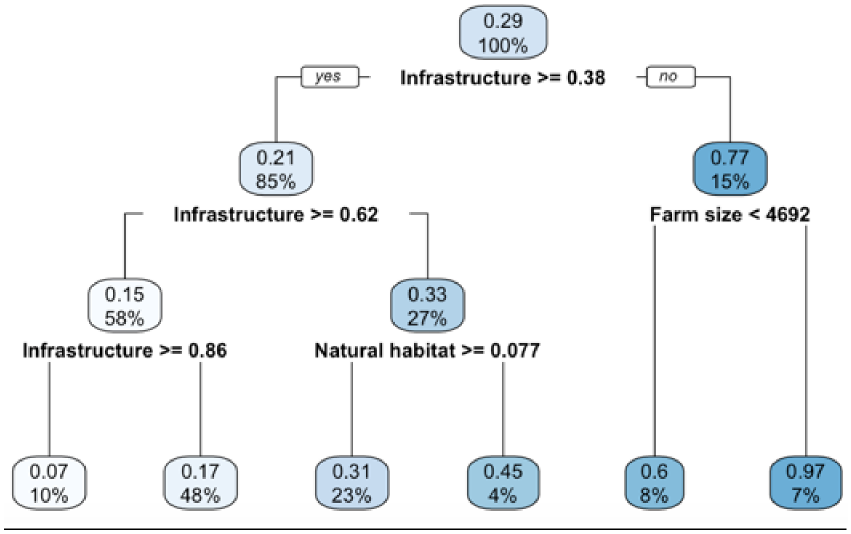


Figure 2. CART Analysis of explanatory variables (proportion of infrastructure, non-crop area, and overall size) predicting the proportion of production area. When conditions of significance are met, the graph branches and lists the value of the explanatory variable.

3.2. On-farm management practices

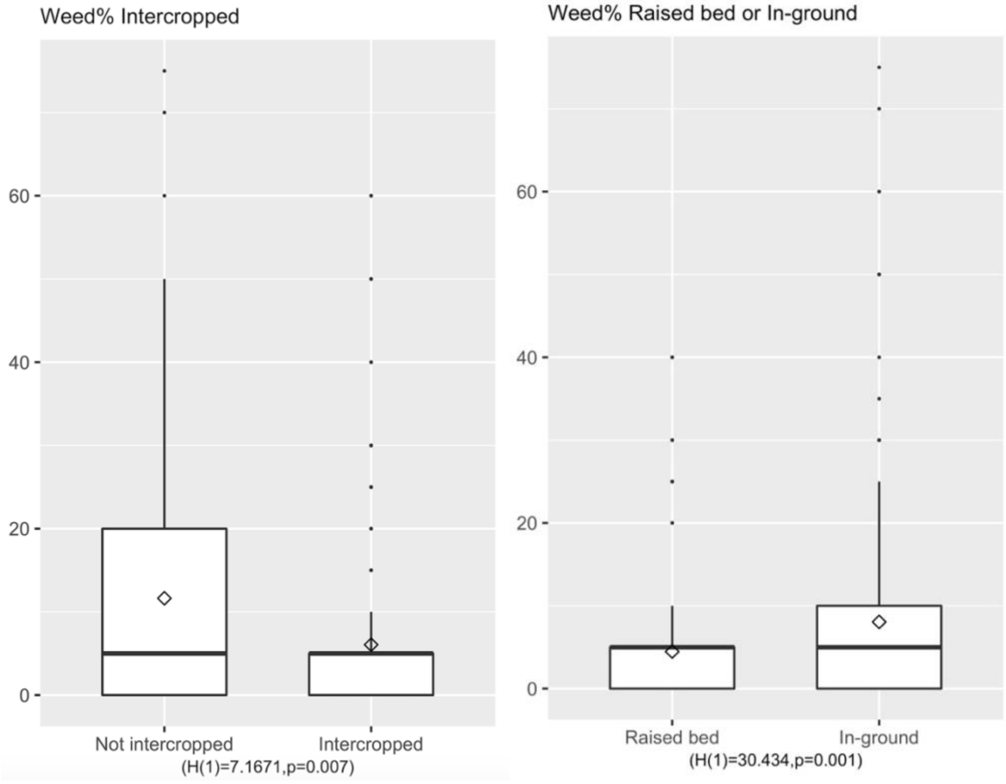
Survey results and ground-proofing indicate that agroecological management practices have been widely adopted throughout East Bay urban farms and gardens. Almost all farms assessed incorporated inter-cropping as well as cover cropping and applied compost. Indexed management practices (crop rotations, cover crops, intercropping, compost, manure, and fish emulsion applications, on-site composting, no-till practices, and whether or not plots were mulched) when compared with mean estimated productivity, weed density, and overall crop biodiversity did not have significant effect.

3.2.1. Crop and non-crop diversity

Crop biodiversity was measured both using quadrats and transects to better capture overall farm crop diversity. Mean quadrat diversity (n=933) was 2.97 crops/m^2. Eight-meter transect count averaged 10.28 crops. Management intensity as indexed by measured agroecological practices did have a weak correlation with crop biodiversity, but this is not reported as crop biodiversity was largely a mechanism of crop selection by urban farmers. Crop biodiversity and management intensity are interesting in the sense that crop biodiversity on-site may be a proxy for agroecological management practices.

3.2.2. Weed occurrence

Overall weed coverage per quadrat was measured over two years. Average weed coverage in quadrats was 7%. Broadleaf weeds were found in greater proportion than grass weeds. Approximately 40% of sampled quadrats had no weeds (Table 3). A non-parametric Kruskal-Wallis Test (see Appendix A4) revealed that there was a statistically significant difference in weed percentages per unit of area between intercropped and not intercropped quadrats (H(1)=7.1671,p=0.007), and raised bed and in-ground production (H(1)=30.434,p=0.001) (Figure 3.).



(**a**) (**b**)

Figure 3. percent coverage of weeds in intercropped beds and in-ground or bed production.

Table 3. Weed coverage in production areas.

Weeds (%/m^2), ***N=933***	
Average weed coverage (m^2) - all quadrats	7%
Quadrats with only broadleaf weeds	31%
Quadrats with only grass weeds	8%
Quadrats with mixed weed (both broadleaf and grass weeds)	21%
No weeds present	40%

3.2.3. Management practices and estimated productivity

Estimated productivity was calculated per square meter quadrat (n=933) at twenty urban agriculture sites. Using yield estimates per plant/quadrat, and total farm production space, we estimate that sampled urban farms produce 7.14kg/square meter (Figure 3.). As productivity estimates were not measures of true production, we used actual yields from two large urban farms that recorded total seasonal yield and divided that number by their overall production area.

Our estimates closely matched comparative true yields (8.6kg/m^2) by weight and area of production on these two farms.

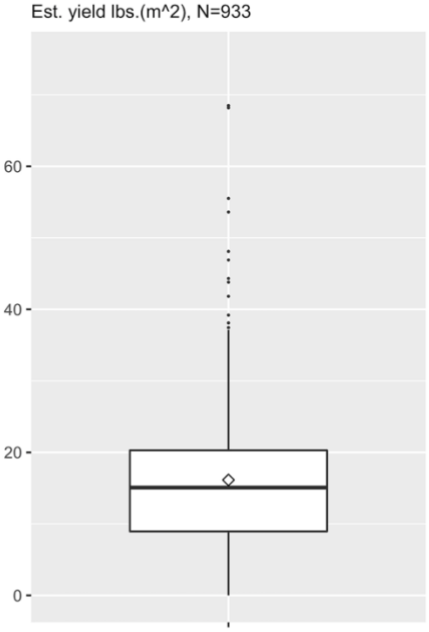


Figure 3. Yield estimates in lbs. per square meter.

3.2.4. Disposition of Harvest

Aggregated survey data indicate that harvests are distributed in the following proportions: Sixty-nine percent of harvests go to the farmers' families or the community surrounding the farm who are affiliated or familiar with the farm operations. Twenty-one percent of harvests go to farmers markets or Community Supported Agriculture (CSAs) supporting the organizations (or members) economically. Ten percent of the harvest goes to organizations that are directly helping vulnerable populations (i.e., shelters and community kitchens).

4. Discussion

Developing a better understanding of the agroecological elements of urban farms will be an important topic in an increasingly urbanized world. Previous analysis suggests that worldwide urban food production can significantly impact global food requirements (Clinton et al., 2018). However, as urban populations grow, urban land becomes increasingly valuable, and the "highest and best use" of vacant urban land may limit the implementation of UA. Production capability, impacts on local food security, and the overall economic efficacy of UA will be crucial in promoting and prioritizing it in future and current urban food systems (Horst et al., 2017; A. Siegner et al., 2018). Further, developing a better understanding of the multi-functionality of UA, including the social, economic, and ecological benefits these systems provide, can better help policymakers and urban planners bolster UA, acknowledging its utility and benefit in the built environment. Understanding spatial composition trends, management practices, and production potential are important and understudied topics that contribute to our understanding of urban farms form and function. This research provides data and context that may influence future discussions regarding the viability and efficacy of UA.

4.1. Spatial composition and potential productivity of urban farms

A deeper understanding of UA production capabilities, especially in the context of on-farm land use, is an important topic when questioning the efficacy of urban food production on high-value urban land. Urban land cycles are largely dependent on rent-seeking and attempt to exploit rent-gaps for profit by landowners and developers (N. Smith, 1996). Developing under-used urban land is often very profitable, counter to UA operations. High land values consistently challenge urban agriculture systems in the context of "highest and best" use - the concept that land-use should always create the most profit. Urban farms are consistently put in a dilemma; they must justify their existence in the context of production. However, if they are not generating substantial profits, their implementation on high-value urban land will always be questioned.

Urban farms also suffer from a fundamental misalignment with "highest and best use" objectives; previously published survey data indicate that urban farm goals are often focused on social goods and food security (J. Arnold & Rogé, 2018). Generating profits is often a tertiary goal at best. Despite this misalignment and lack of financial support, estimated yields per unit area are high, with approximately 7.14kg/square meter of fresh vegetables being grown. Urban farms also significantly impact local food security, with ~69% of on-farm production going to the local community.

Our findings indicate that increasing overall production capacity in UA, an important consideration in urban land use, can be linked to on-farm land use. Despite high yields per unit of area, our on-farm spatial analysis found that an average of only ~32% of available area is being utilized for production. With land being such a limiting factor of UA adoption, we found that UA sites may not be maximizing potential production area. These findings indicate that overall urban farm size is not a limiting factor to increased production. Two possible explanations may influence underutilization of production area. Firstly, these farms often exist on volunteer labor and often lack consistent funding to pay farm managers and employees (J. Arnold & Rogé, 2018). Investment, infrastructure, and labor may be limiting full production potential. Moreover, these spaces are serving residents more than markets. If local food needs are met there may be less incentive to put additional land into production. Conversely, spatial composition, especially in the context of production area, may be influenced by management practices. The three farms with the highest proportion of production area all utilized in-ground management practices. Contrariwise, UA sites with the most significant proportion of infrastructure (and minimal production area) all utilized raised-beds in their production systems.

Raised-bed production is often linked to concerns about soil health or security of tenure. Raised-bed production can help mitigate potential soil contamination issues, often found in UA. This production practice is also modular and can be broken down and moved in cases of insecure tenure. In summary, UA production is not limited by yields per unit of area but more explicitly linked to social-ecological factors that prohibit the full implementation of long-term, in-ground production systems.

4.2. Agroecological management practices on urban farms

We found that implementation of sustainable farming practices is widespread among urban farmers and practiced across measured sites. Intercropping, cover-cropping, and soil building practices are common and often practiced simultaneously (Table 2.). Adoption of sustainable farming practices may be in response to abiotic and ecological challenges faced by converting impacted urban land into productive farms. Crop rotations, cover-cropping, mulching, and manure and compost application were often cited during interviews with farm managers as strategies to remediate impacted urban soils. Management practices were also frequently cited as strategies used in response to pest and weed pressures. Weeds were prevalent in all measured sites, but broadleaf weeds were most pervasive and were especially abundant for in-ground production systems as opposed to raised beds. Average weed coverage in quadrats was reduced by the implementation of intercropping. These results have important analogs to findings in rural agricultural systems and show that these practices can be implemented at small scales in novel urban agroecosystems.

5. Conclusions

Our findings help us better understand urban farm spatial composition and management practices. We found that urban farms are diverse in spatial composition, have adopted a broad spectrum of agroecological management practices, are highly productive, and directly impact local food security. We were unable to link specific practices to increased production per area unit, but we found that intercropping can decrease weed occurrence. Underutilization of available farm area for production was prevalent. We propose that production limits are linked to social and economic factors that prohibit urban farms from developing high-yielding, in-ground production systems. Our findings on UA form and function give us a better understanding of how urban farms function in the landscape. However, they are often under-supported and often suffer from insecure tenure, limiting their potential impact (J. Arnold & Rogé, 2018; Daftary-Steel et al., 2015). Despite these limitations, UA systems continue to impact local food security and provide a myriad of social goods to local communities. Changing perspectives of how UA systems function and their social and economic benefits will be of great importance for their continued existence in high-cost cities.

Chapter 2: Local and landscape effects to biological controls in urban agriculture – a Review

Abstract

Urban agriculture is widely practiced throughout the world. Urban agriculture practitioners have diverse motivations and circumstances, but one problem is ubiquitous across all regions: insect pests. Many urban farmers and gardeners either choose to, or are required to forego, the use of chemical controls for pest outbreaks because of costs, overspray in populated areas, public health, and environmental concerns. An alternative form of pest control is conservation biological control (CBC)—a form of ecological pest management—that can reduce the severity of pest outbreaks and crop damage. Urban farmers relying on CBC often assume that diversification practices similar to those used in rural farms may reduce insect pest populations and increase populations of beneficial insects, yet these management practices may be inappropriate for applications in fragmented urban environments. In this review, we assess urban CBC research and provide a synthesis for urban agriculture practitioners. Our findings indicate that local and landscape factors differentially affect insect pests and beneficial arthropods across the reviewed studies, and we identify several on-farm practices that can be implemented to increase biological control in urban agriculture.

1. Introduction

Urban agriculture (UA) is defined as agricultural production within urban areas managed by urban residents (henceforth "urban farmers") including home gardens, market farms, orchards, and often, animal rearing (Zezza & Tasciotti, 2010). The popularity of UA has expanded in cities around the world (Mok et al., 2014). The American Gardening Association reported a 34% increase in new urban farms between 2007–2011 and identified over 8500 operating urban farms and gardens in 38 US cities (Lawson & Drake, 2013). The realized and potential benefits of UA are far-reaching; recent estimates claim UA could annually contribute $80–160 billion (US) in food production, nitrogen fixation, energy savings, pollination, climate regulation, soil formation, and biological control of pests (Clinton et al., 2018). There are innumerable variations of UA worldwide, with various on-farm compositions, each situated in their own agronomic and geopolitical context. This review does not attempt to be inclusive of all variations of UA, but to focus on the ecological management of crop pests and assess the current state of research of biological control in urban agriculture—an ecosystem service with an estimated value of $1.12 billion (US) (Clinton et al., 2018).

1.1 Pests, natural enemies and pest control in urban agriculture

One of the most significant challenges reported by urban farmers is crop pests (Gregory et al., 2016; Oberholtzer et al., 2014). Pests in UA are ubiquitous, and characteristics of urban areas can make pests particularly damaging and difficult to control. Herbivorous insect populations have been reported to decrease in diversity but increase in abundance in urban areas (Faeth et al., 2011), and pest outbreaks are linked to factors endemic to urbanization—habitat fragmentation and disturbance (Heimpel & Mills, 2017). Other unique features of urban areas such as vegetation maintained year-round, nutrient-stressed perennials, and higher temperatures from the urban heat island effect can also increase pest density and/or the severity of pest damage (Dale & Frank, 2014; Faeth et al., 2011; Meineke et al., 2013; Turrini et al., 2016). Despite these challenges, many urban farmers choose not to use pesticides for public and environmental health reasons (Oberholtzer et al., 2014), instead using ecological pest management practices (Altieri et al., 1999). Moreover, because of re-entry and pre-harvest intervals, many of the more effective pesticides cannot be used on typical urban farms where multiple plant species are adjacent, and the farm is visited daily by UA practitioners. In contrast, conservation biological control (CBC) uses practices that are commensurate with many UA practices and limitations by employing habitat manipulation to provision resources that can support "natural enemy" arthropods to improve pest suppression (Heimpel & Mills, 2017).

Diverse management practices such as crop rotations, intercropping, increased plant species richness, and incorporation on non-crop habitats contribute to high spatial and temporal diversity in UA systems (Altieri et al., 2016; Clarke & Jenerette, 2015; Lin et al., 2015; Loram et al., 2008; Taylor et al., 2016), but information about how these manipulations affect ecosystem function, especially CBC, is inadequate in comparison to research in rural farms. For example, numerous studies have reported that habitat manipulation and diversification of the surrounding landscape and on-farm biodiversity have been effective at increasing beneficial insect richness, abundance, and biological control on more rural farmscapes (Altieri, 1999; Bianchi et al., 2006; Chaplin-Kramer et al., 2011; Letourneau et al., 2011). At the local-scale incorporation of non-crop perennials, floral resources, and crop rotations within farms (Crowder & Jabbour, 2014; Landis et al., 2000; Rebek et al., 2006; Rusch et al., 2013), and at the landscape-scale, greater proportions of natural vegetation, non-crop land, and landscape heterogeneity surrounding rural farms have proven to promote the biological control of pests (Tscharntke et al., 2005). Adding floral resource additions, crop rotations, and ground cover management practices including mulching, and soil amendments such as compost additions can provide benefits such as alternative food sources and habitats necessary for maintaining consistently high natural enemy populations and increased rates of biological control over time and space (Landis et al., 2000; Rusch et al., 2013; Tamburini et al., 2016). Further investigating these practices in UA can help provide ecologically based, cost-effective interventions to reduce crop damage from insect and mite pests, thereby increasing local food security.

1.2 Research and Extension in Context of Urban Agriculture

UA practitioners often adopt agroecological practices that include local habitat diversification, but there are few studies that document whether the impacts of diversification on small urban farms are similar to more rural, larger agricultural systems that are not subject to affects unique to UA systems, including urban microclimates, reduced species diversity, and landscape-scale characteristics. A growing field of study in urban biological control has sought to fill this research gap. Some limited work has shown how local- to landscape-scale effects often vary by taxa (Tamburini et al., 2016), and by the type of crop damage, ranging from chewing herbivory to fungal and bacterial disease. Of importance to urban farmers, multiple on-farm practices have been identified that may be implemented to increase CBC in urban farms (Philpott & Bichier, 2017). To our knowledge, this work has not yet been gathered and synthesized, making it difficult to translate research into practice. Here, we review and summarize relationships between local farm and surrounding landscape effects in UA on pest and natural enemy populations, as well as on the resulting levels of biological control (ecosystem services). Our literature review focuses on four questions for UA systems: Which local on-farm practices and off-farm landscape factors affect (1) insect and mite pest populations and their crop damage; (2) natural enemy biodiversity (abundance, species richness, community composition); (3) ecosystem services through increased biological control; and (4) which practices can be recommended to urban farmers to promote CBC?

2. Methods

We searched for peer-reviewed literature, published before February 2019, that measured natural enemy and insect pest richness, abundance, and rates of predation and parasitism in UA systems. We further focused the review on intra-urban studies (comparison of urban farms) that measured differences in on-farm composition, practices, and surrounding off-farm landscape attributes to measures of insect abundance, richness, and community composition. We excluded studies that either focused on taxa that do not provide regulating ecosystem services relevant to CBC (e.g., some Lepidoptera, non-parasitoid Apoidea, and Orthoptera), or compared pest or natural enemy abundance, richness and composition between urban and rural green spaces, farms, or gardens. We did this because these measures do not explicitly focus on UA or local on-farm predictors of arthropods. In some cases, we did include urban-to-rural studies if a subset of the samples met the intra-urban requirement. For these studies, we excluded the reported findings from rural or natural landscapes.

The review protocol followed the PRISMA systematic review framework and the methodologies described in Pullin 2006 (Pullin & Stewart, 2006). We searched three databases including Web of Science, the United States Department of Agriculture National Agricultural Library database (AGRICOLA), and the National Center for Biotechnology Information (Pubmed), using search terms that are common in the CBC literature: “Biological control,” “Herbivore,” “Pest,” “Parasitism,” “Natural enemies,” and “Parasitoid.” These terms were paired with “Urban agriculture” and “Urbanization.” Search terms were applied to titles, abstracts, and keywords. Our search protocol identified 675 peer-reviewed publications using this methodology. We removed all duplicates and reviewed the remaining articles (N = 582) for relevance. From these, we identified 15 articles that met our protocol criteria and were selected for review (Table S1).

For each publication, we collected information on authors, title, site location(s), site sample number, land type (e.g., garden, park, etc.), sampling period, methodology, and taxa assessed. We then recorded the statistically significant effects of 16 explanatory variables common among studies for species richness, abundance (for pests and natural enemies), and levels of ecosystem services through biological controls (Table 1). To further identify explanatory variables, and to align variables with reviewed literature, we categorize variables as “local factors” or “landscape factors.” Local factors were defined as biotic and abiotic features of the local agroecosystem (e.g., vegetation and ground cover that are manipulated through specific practices at the farm scale), and landscape factors were defined as features of the surrounding landscape (e.g., land use composition and land use type diversity). For each explanatory variable, we counted the number of reportable results (Table 1). Some explanatory variable measures, such as local or landscape factors combined into an index value, or measures that were not clearly defined were categorized as "landscape cover" or "structural diversity" (M. H. Egerer et al., 2017).

Table 1. Explanatory variables from local and landscape effects that were categorized from the literature review of 15 articles on their positive/increasing (+) or negative/decreasing (-) impact on species Abundance (A) and Richness (R) of parasitoids, predators, herbivores, and ecosystem services predation (P^r), and parasitism (P^a).

Explanatory Variables	Parasitoids				Predators				Herbivorous Taxa				Predation/Parasitism			
	A+	R+	A-	R-	A+	R+	A-	R-	A+	R+	A-	R-	P^r+	P^a+	P^r-	P^a-
Landscape Effects																
Impervious surface (% high)	2		1	1	3	3	2	2		1	1	1	2		1	
Impervious surface (% low)	1				3	2				1						
Proximity to agriculture	1														1	
TOTAL	4	0	1	1	6	5	2	2	0	2	1	1	2	0	2	0
Local Effects																
Garden size (Large)	2	1			1	1									1	
Garden size (Small)					2							1	1			
Host density									1							1
More perennial					1					2			3		1	
Less perennial	1				1											
Height of perennial						1				1						
Plant species richness (High)	1	1	1		1	2									1	
Plant species richness (Low)															1	
Structurally diverse		1^{1}			2^{1}	1^{1}				1						
Mulch	2					2									1	
Leaf litter													2			
Landscape cover (increased)						1					1		2			
Landscape cover (decreased)					1			1					2			
Canopy cover			1								1					
Floral abundance	1	1^{2}			3						1					
Floral richness	1	1^{2}														
Garden age (older)										1		1				
Height of herbaceous cover		1														
TOTAL	9	6	2	0	1	8	0	1	1	5	3	2	9	0	5	1

[1]: VCI: Vegetational Complexity index (VCI) as measured in Egerer et al. (2017) is categorized as Structural Diversity.

[2]: Floral: Floral additions in Egerer et al. (2018) are measured as floral abundance and richness.

4. Results

The selected articles were published between 2006–2018; most studies were from Europe or the Northern Hemisphere, with only one of the studies occurring in the Southern Hemisphere (Morales et al., 2018). Studies varied by level of taxonomic classification, with most identifying arthropod taxa to morphospecies or family/superfamily. Parasitic Hymenoptera and predaceous Coleoptera including ground beetles (Carabidae) and ladybird beetles (Coccinellidae) were the most studied taxa (Figure 1). Studies generally did not consider life history strategy or feeding guild; for example, whether arthropods were generalist or specialist in their prey or host selection. When studies explored more than one taxa, we only included those results that were comparable to other reviewed publications.

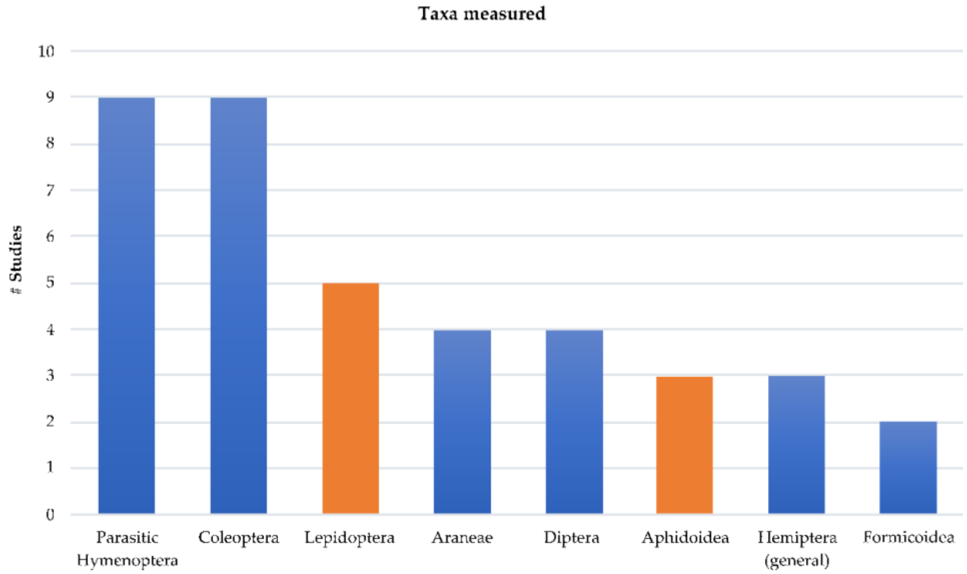


Figure 1. Number of taxa studied in the reviewed published literature. Height of the bar represents the number of studies for each taxa. Blue coloration represents predator or parasitoid groups, orange represents pest groups considered in each study.

3.1 Effects of Local on-Farm Management on Taxa

3.1.1 Herbivorous Insect Pests

In four publications, herbivorous taxa were studied that are UA crop pests, but few local factors were presented that explained increases in pest abundance or richness. Moreover, studies often showed inconsistent results, and the only factor repeatedly associated with increased pest richness was increased perennial richness and abundance. Structural diversity of vegetation, host plant density, garden age and soil moisture were also identified as factors affecting herbivore richness and abundance, but these relationships were only measured as significant once.

3.1.2 Natural Enemies

Local factors positively affected parasitoid and predator abundance and richness in thirteen of fifteen reviewed studies, with only 7% of the reported results showing negative effects on natural enemy abundance and richness. Important local factors that positively affected natural enemy populations included increased floral abundance and richness, increased mulch and leaf litter cover, larger garden size, high plant species richness, more perennials, and increased structural diversity. Garden size was the only factor that differed between predator and parasitoid taxa, with larger gardens positively affecting parasitoid populations and smaller gardens positively affecting predator (e.g., beetle) abundance.

3.2 Effects of Surrounding Landscape on Taxa

The amount of urbanization surrounding UA sites was the most frequently measured landscape factor (n = 14 of 15), and 71% of the studies reported a significant effect on arthropod populations. However, the direction and magnitude of the relationships were highly variable across arthropod taxa (Table 1).

3.2.1 Herbivore Insect Pests

Landscape factors had little effect on herbivorous taxa, only three studies found positive relationships between landscape factors and herbivore abundance and richness. Richness was positively affected in both low and high rates of surrounding impervious surface, and both richness and abundance were negatively affected by high rates of impervious surface.

3.2.2 Natural Enemies

Both higher and lower amounts of impervious surface (e.g., concrete roads and buildings) surrounding urban farms positively affected natural enemy populations. Of the nineteen reported results in reviewed studies associated to natural enemies and landscape factors, 70% positively affected natural enemy richness and abundance. However, all negative effects (30%) were associated with high levels of impervious surface (e.g., asphalt). Predator taxa were more strongly affected by a high impervious surface, accounting for 66% of negative effects. Parasitoid Hymenoptera were also affected by low and high rates of impervious surface similarly.

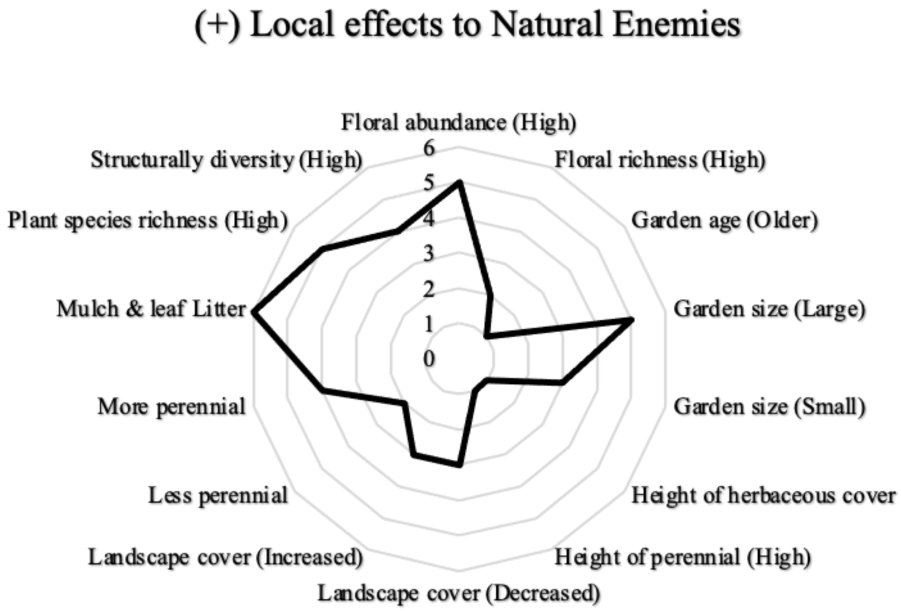


Figure 2. Numbers along the center y-axis represent the number of reportable results in the reviewed literature that indicate a positive effect to natural enemy richness, abundance, and rates of biological control. Reported results are correlated with explanatory variables (local factors) listed on the exterior of the radar chart.

3.3 Local and Landscape Effects on Conservation Biological Control

Local factors were important for explaining levels of ecosystem services through the biological control of arthropod pests (Figure 2). Nine studies recorded higher rates of predation associated with a local factor. Increased predation rates were associated with higher perennial abundance, leaf litter and landscape cover, and smaller gardens. Negative effects on predation included larger gardens, plant species richness (both high and low) and mulch. No studies found a local factor associated with positive parasitism rates. For landscape factors, urban land cover at varying spatial scales was associated with predation rates in four studies, including high and low rates of urban cover and proximity to agriculture land use in the surrounding landscape.

5. Discussion

We reviewed UA literature to assess how local on-farm management practices and surrounding off-farm landscape features affect herbivorous insect pests, arthropod natural enemies, and measures of conservation biological control. This is a first attempt to synthesize the growing number of case studies in this field.

We found that local and landscape factors differentially affect insect pests and their natural enemies, as well as ecosystem services received through biological control. Local on-farm diversification and management most commonly affected natural enemy species richness, abundance, and ecosystem services with (78%) of reported results showing positive impacts.

Relationships between measures of arthropod diversity and impervious urban land cover at the landscape scale are inconsistent, as they have both negative and positive effects on arthropod populations (Mace-Hill, 2015). Some reviewed studies found parasitoid abundance increased, but richness decreased with urban landscape cover (Burks & Philpott, 2017), or that these relationships for predators are differential across taxa, region, and landscape scale (M. H. Egerer et al., 2017; M. H. Egerer, Liere, Lin, et al., 2018). The differences across taxa, region, and surrounding urban landscape composition are all important considerations.

Arthropod pests were generally unaffected by local or landscape scale factors. However, insect pests were the least commonly measured taxa across these studies. Only two reviewed studies focused on intra-urban local and landscape herbivorous pest effects, and most studies did not assess relationships between insect pests and crop damage (M. H. Egerer, Liere, Lin, et al., 2018; Lowenstein & Minor, 2018). Rates of parasitism were also unaffected by local and landscape factors, even though parasitoids are prevalent in urban gardens (Burkman & Gardiner, 2014; Burks & Philpott, 2017). Similarly, urban-to-rural studies report that parasitic Hymenoptera may be somewhat resistant to landscape-scale habitat fragmentation in larger non-garden habitat patches (Christie & Hochuli, 2009). However, in more urbanized landscapes with smaller habitat patches, landscape fragmentation has negative effects on Hymenoptera species diversity (Bennett & Gratton, 2012).

Our review identified gaps in UA CBC-related research, particularly on the topics of methodology and geographic breadth. The key methodological issues that we found in the literature include: (1) lack of measured temporal effects; (2) inconsistent sampling techniques across studies; (3) coarse taxonomic identification and biodiversity metrics of focal taxa; and (4) difficulty in accessing sufficient landscape data. Only three of the reviewed studies measured temporal effects (M. Egerer et al., 2018; M. H. Egerer, Liere, Lin, et al., 2018; Lowenstein & Minor, 2018), and the average sampling period was only 22 weeks. Clearly, more extensive year-round sampling is needed to account for possible temporal changes between seasons. To this point, local climate measures were rarely reported; only a third of studies measured temperature, and none measured wind speed or humidity. These local abiotic climate-related factors should be considered as climate change will increasingly impact urban arthropods in the coming decades.

It is important to consider methods of insect sampling and units of ecosystem function in CBC research. Often the goal of UA studies is to better understand functionally important species distributions in fragmented landscapes with implications for agricultural ecosystem functioning. While measuring the richness and abundance of insects is an essential step to understand species distributions, it does not account for functional effects of biodiversity that are of use to UA practitioners. Nineteen of the studies used pan traps or sticky traps, standard but often superficial methods in insect population studies. These sampling methods can be too broad when investigating biological controls (Doxon et al., 2011; M. H. Egerer, Liere, Bichier, et al., 2018; McCravy, 2018). It would be useful to measure the actual rates of prey consumption, for example, by using exclusion and sentinel prey in relation to natural enemy presence, or by rearing parasitized insects. Emerging technologies such as molecular gut content analysis of predators using DNA-based prey assays are an effective method to link predator to pest (Chisholm et al., 2014). While most studies included multiple methodologies, it would be useful to include more standardization in UA field sampling protocols for biodiversity and biological control to facilitate future meta-analysis.

With regard to biodiversity metrics in UA CBC research, most of the reviewed studies offer only a coarse overview of species identification, especially for parasitic Hymenoptera, which are often only identified to superfamily. Though genus- and species-level identification are time-intensive and require skilled labor, species- or genus-level data is necessary to better investigate species-effects on CBC. This is particularly important because many of the parasitic Hymenoptera in the reviewed studies are aggregated as "beneficial," but many are hyperparasitoids, or are potentially parasitoids of other natural enemy predators (Burks & Philpott, 2017; Lagucki et al., 2017; Sperling & Lortie, 2010). More research is needed on UA pests because insect pests are the least measured taxa across studies, and most studies do not assess relationships between insect pests and crop damage and yield. Only two reviewed studies focused on intra-urban local and landscape herbivorous pest effects (M. H. Egerer, Liere, Lin, et al., 2018; Lowenstein & Minor, 2018).

Additional challenges exist in UA landscape studies, notably the availability of fine-scale landscape data. Urban ecologists have been limited in their access to geographical data at a scale smaller than 30 m. Many studies use the US National Landscape Cover Database, which includes a measure of the impervious surface, but the scale is inappropriate for complex urban environments particularly, when considering effects on arthropods that respond to habitat heterogeneity at much smaller spatial scales (Qian et al., 2015). Ground-proofing landscape composition can necessitate consistent access to a private property which can be challenging. Alternative methodologies have been proposed such as aerial drones with high-resolution cameras, but limits to drone flight plans in residential areas or excluded flight space make flights difficult. We also found a strong bias towards UA studies in North America and Europe. It is unclear as to whether this bias is a relic of the database search itself or there is a distinct lack of literature available.

6. Conclusions

The mixed results presented in the literature reviewed, and their varied measurements and foci, suggests that accessing knowledge about urban CBC in UA as a layperson can be challenging. Urban extension services can use this aggregated information to bridge divides between research and practitioners, and to influence on-farm practices for increased CBC and agricultural sustainability. Improving UA sustainability through CBC practices will also require better support, bolstering, and expansion of UA extension services as most agricultural extension is biased to rural systems. Throughout this work, we have identified several management practices that can guide urban farmers, extension agents, urban planners, and policymakers. While urban farmers cannot necessarily control for landscape features in urban areas, they can implement practices that affect local-scale vegetation complexity such as increasing plant species richness, floral provisioning, incorporation of more perennials, and increased ground cover heterogeneity. Maintaining biodiversity at multiple scales of the agroecosystem through urban farm management supports principles of agroecology and can build sustainability and increase ecosystem function over time.

Transition statement for previously published literature

Chapters 1&2 are previously published by the author. These important preliminary chapters contextualize the reader's understanding of urban agroecosystem composition and the effects of on- and off-farm characteristics on biological control services in urban agroecosystems. These chapters are critical for understanding the research performed in chapter 3&4 but also stand alone as important findings regarding urban agroecosystem function.

Now, in chapter 3&4, I will share the research results on local and landscape effects on parasitic Hymenoptera and the Cabbage aphid by testing two fundamental theories in agroecological pest management: the enemies hypothesis and the floral nectar provisioning hypothesis in urban agroecosystems. Chapter 1: *On-Farm Spatial Composition, Management Practices, and Estimated Productivity of Urban Farms in the San Francisco Bay Area* contextualizes the agroecosystem composition in the study landscapes. Chapter 1 supports and builds context around urban farm characteristics, practices, and composition that become important in understanding the overall characteristics that impact the response variables important to agroecological pest management explored in chapters 3&4.

In Chapter 2: *Local And Landscape effects to Biological controls in urban agriculture - A Review*, I introduce the reader to the most up-to-date literature on biological control services in urban agroecosystems available during this dissertation's research design and data collection. These findings place the results in chapters 3&4 in the greater context of research, both justifying the merits of the methodologies and detailing the complexity of urban agroecology.

Chapter 3: Biological control services from parasitic Hymenoptera in urban agriculture

Abstract

Urban agriculture is practiced in spatially fragmented landscapes with unique characteristics that can impact species occurrence in time and space. As a result, biological control services, an ecosystem service from naturally occurring arthropod natural enemies, can be negatively impacted. Many urban farms forgo pesticides and utilize agroecological pest management strategies that rely on natural enemies to help regulate pest populations. Understanding how these enemies are affected by landscape composition and on-farm management practices is critical to understanding agroecological pest management in UA and furthering our understanding of landscape-mediated population dynamics. Over two growing seasons, we sampled brassica crops in urban agriculture sites occurring on a spectrum of surrounding landscape imperviousness, spatial composition, size, and management practices to better understand parasitic Hymenoptera abundance, richness, and parasitism rates on the common cabbage aphid (*Brevicoryne brassicae*). We found that on-farm agroecological pest management practices such as mulch coverage, floral richness, and overall crop plant richness impacted parasitic Hymenoptera abundance. Larger proportions of on-farm non-crop area increased parasitoid abundance on urban farms. Aphid parasitism increased in relation to on-farm management practices, including increased crop plant richness. These findings add to a growing understanding of urban agroecosystem function and support the enemies hypothesis in urban agroecosystems.

1. Introduction

Crop pests in urban landscapes can be challenging to control, and they can have a disproportionate impact on the smaller crop sizes common to urban farms. Herbivorous insect populations in urban areas can persist for more extended periods, have increased fecundity, and can even be larger (Bowler et al., 2010; Dale & Frank, 2014, 2018; Douglas & Tooker, 2015; Korányi et al., 2022; Parsons & Frank, 2019; Tooker & Hanks, 2000). Undermanaged or neglected urban landscapes can exacerbate pest issues. Irregular irrigation, application of fertilizers or pesticides, and higher levels of air pollution can induce plant stress or vigor, creating favorable conditions and refuges for herbivorous pests that can emigrate to urban farms and gardens (Galway et al., 1997; Raupp et al., 2010). Urban agriculture (UA) is often practiced without pesticides for health and environmental reasons despite these challenges. Instead, farmers find themselves relying on time and labor-intensive cultural and mechanical practices for pest management. Consequently, urban farmers have shown great interest in agroecological pest management (APM), a proactive ecosystem services based approach that aims to reduce pest abundance and crop damage by increasing natural enemy populations through agroecological practices (Altieri & Nicholls, 2000; Deguine et al., 2009). For example, increasing vegetative complexity, implementing soil conservation practices, and introducing floral resources for nectarivorous natural enemies. These practices increase the amount of shelter, nectar, and pollen resources on urban farms, increasing natural enemy populations, resulting in increased biological control services (Landis et al., 2000).

1.1 Rural on- and off-farm diversification effects

In rural agroecosystems, APM practices, landscape effects, and conservation biological control have been widely studied (Chaplin-Kramer et al., 2011; Letourneau et al., 2011). Meta-analyses have found that on-farm management practices such as intercropping, crop rotations, and increased structural diversity increase natural enemies' abundance, diversity, and ability to regulate pest populations (Letourneau et al., 2011). Increased landscape diversity surrounding rural agroecosystems has been shown to mediate arthropod diversity and abundance, with natural enemies showing a positive response to increased landscape complexity (Chaplin-Kramer et al., 2011). The enemies hypothesis states that increased structural complexity should increase natural enemy abundance, diversity, and associated ecosystem services (Andow, 1991; Pimentel, 1961; Root, 1973; Tahvanainen & Root, 1972). This hypothesis has been investigated and questioned in agroecosystem management, with varying results at different spatial and temporal scales, most often in rural contexts. Ostensibly, diversification effects observed in rural agroecosystems should be observed in their urban counterparts. However, the effects of diversification on biological control services and APM in urban agroecosystems regarding this hypothesis are still being explored, especially in how natural enemies are affected by landscape factors such as fragmentation and isolation, common in urban landscapes (J. E. Arnold et al., 2019).

1.2 Urban fragmentation effects

The extent of fragmentation effects on organisms in urban environments, and related ES, has been a persistent question, especially in urban agroecosystem management (Burkman & Gardiner, 2014). Roads, parking lots, and buildings increase impervious surfaces, fragmenting and reducing greenspace connectivity and impacting the quality and area of suitable habitat (Driscoll et al., 2013; Forman & Godron, 1981). The reduction of available and appropriate habitats for urban flora and fauna decreases metapopulation connectivity and drives a decline in urban species diversity, selecting for more disturbance tolerant species and increasing the chance of localized extinctions (Alberti, 2005; Driscoll et al., 2013; Faeth et al., 2011; Fahrig & Nuttle, 2005; Kennedy et al., 2011). Existing literature on the effects of urbanization on species occurrence, abundance, and diversity often relies on urban-rural gradient studies (Bennett & Gratton, 2012; Burkman & Gardiner, 2014; Turrini et al., 2016). These studies generally find that increased urbanization decreases the diversity of organisms (Alberti, 2010; Faeth et al., 2011). Confirming these findings are an abundance of patch-matrix literature suggesting that the quality of the habitat patch itself, its size, and the composition of the matrix surrounding it are determining factors for species occurrence in fragmented landscapes (Driscoll et al., 2013; Hanski, 1998; Kennedy et al., 2011; Prugh et al., 2008). Specific to UA, higher imperviousness surrounding urban farms has been related to decreased parasitoid abundance and richness (Burks & Philpott, 2017; Morales et al., 2018), decreased predator abundance and richness (M. Egerer et al., 2018; M. H. Egerer et al., 2017; Mace-Hill, 2015), and even decreased predation on sentinel prey (Philpott et al., 2014).

1.3 Management effects on natural enemies

While landscape effects in UA have also been shown to increase abundance (M. H. Egerer et al., 2017; Lagucki et al., 2017; Sperling & Lortie, 2010), and diversity of natural enemies (Burks & Philpott, 2017; M. H. Egerer et al., 2017; Otoshi et al., 2015), the composition of the overall matrix in urban areas is often outside of the scope of management of urban farmers. However, patch quality is easily manipulated through management practices. Abundance of perennials, height of herbaceous cover, and amount of semi-natural or non-crop area on urban farms have been measured in UA and shown to positively impact a wide diversity of natural enemies (J. E. Arnold et al., 2019). Area of ground cover, especially mulch cover, has been correlated with increased natural enemy abundance (Burks & Philpott, 2017; Lagucki et al., 2017; Morales et al., 2018) and richness (Burks & Philpott, 2017; Otoshi et al., 2015). Increased proportions of complex ground covers have been associated with increased rates of prey removal in sentinel prey trials in urban gardens (Philpott et al., 2014). Increased floral abundance and diversity has been shown to increase natural enemy abundance (M. H. Egerer, Liere, Lin, et al., 2018; Lowenstein & Minor, 2018; Mace-Hill, 2015; Morales et al., 2018), and richness (Bennett & Gratton, 2012; M. H. Egerer, Liere, Lin, et al., 2018; Lowenstein & Minor, 2018). Moreover, spatial configuration of working landscapes has become an increasingly important aspect of species occurrence and related biological control services (Fahrig & Nuttle, 2005; Haan et al., 2020). Understanding how landscape and management may affect natural enemy abundance and diversity of natural enemies is an important aspect of effective APM in UA (Kruess & Tscharntke, 1994, 2000; Prugh et al., 2008; Tscharntke et al., 2012).

1.4 Parasitic Hymenoptera

This research continues to build on previous findings from studies of local and landscape effects on biological control services in UA by focusing on parasitic Hymenoptera (PH) in urban agroecosystems. PH are important in provisioning biological control because they utilize arthropod hosts during their juvenile life stages, leading to the termination of hosts. Previous studies focused on UA in the context of local and landscape effects on PH have found both increased and decreased abundance with higher rates of imperviousness between 200m - 500m (Bennett & Gratton, 2012; Burks & Philpott, 2017; M. H. Egerer, Liere, Lin, et al., 2018; Lagucki et al., 2017), and with larger gardens (M. H. Egerer, Liere, Bichier, et al., 2018; Lowenstein & Minor, 2018; Morales et al., 2018). APM practices, including floral provisioning have been shown to increase both abundance and diversity of PH (M. H. Egerer, Liere, Lin, et al., 2018; Lowenstein & Minor, 2018), increased mulch coverage has been shown to increase PH abundance (Burks & Philpott, 2017; Morales et al., 2018), and structural diversity increases PH richness (Burks & Philpott, 2017; M. H. Egerer et al., 2017). Due to the mixed results of pervious findings, especially in the context of potential beneficial affects to APM practices, further research is necessary.

1.5 Research goals

This research focuses on PH-mediated biological control services in brassica cropping systems in UA. Specifically, we focus on intra-urban effects (landscape composition, on-farm spatial composition, and management practices) on the abundance of different taxa of PH and the parasitism of aphids. We attempt to clarify previous findings by focusing solely on brassica cropping systems ubiquitous across urban farming systems in the San Francisco Bay Area, USA. We hypothesize that urban farmers can increase on-farm biological control services by controlling for patch quality through agroecological pest management practices, further supporting evidence for the enemies hypothesis in fragmented landscapes. To test this hypothesis, we investigate whether APM practices (mulch coverage, floral richness, and increased crop richness) significantly affect the abundance of PH in UA. Secondly, we question if surrounding imperviousness and on-farm spatial composition influence biological control services. Lastly, we question whether APM practices impact PH abundance and biological control services to a greater extent than landscape factors.

2. Methods

2.1 Study design

To better understand PH richness and abundance in urban farms and associated biological control services, we conducted an in-situ survey at urban community farms in the East Bay of the San Francisco Bay Area, USA. Eleven farms participated in 2018 and ten farms in 2019. Farms were asked to participate in research based on two factors: 1. farm size, to ensure a comparative sample of small, medium, and large farms, and 2. high or low levels of surrounding impervious surface per the National Landscape Cover Database (NLCD) (see Appendix B1). Landscape factors and APM practices of farms were measured. APM practices included area of non-crop usage (includes all non-crop vegetation), area of production, crop plant abundance (brassica), crop richness, floral richness, and percent of farm surface with complex ground covers including mulch and leaf litter. Landscape factors included percent of impervious surface at 200-, 500-, and 1000-meter radii. Sampling iterations occurred from May to mid-October each year.

On-farm non-crop area was defined as a not actively managed area of the farm occupied by non-crop flora. Farm size in m^2 was calculated through Google Earth Pro and ground-proofed during on-farm spatial measurements. Brassica abundance was determined by counting all brassicas on the farm when sampling occurred. Crop plant richness was determined by eight-meter transects measured perpendicular to garden beds three times during the growing season. Different cultivars of the same species (e.g., kale and broccoli) were counted separately when measuring crop richness. Floral richness was surveyed three times per growing season (early, mid, and late) by completing a comprehensive count of each flowering plant at each survey site. Randomized $4m^2$ quadrats were used to estimate percent of and type of cover (woody mulches or leaf litter). Ground cover quadrats were measured across crop and non-crop areas. Percent of surrounding impervious surface (e.g., pavement, buildings, or other structures) for each farm was measured using the NLCD at 8m resolution (see Appendix B1).

2.2 Sampling methodologies

Collection of PH was accomplished by using an insect vacuum on *Brassica oleracea* cultivars, including broccoli, kale, collards, and tree collards. Each sampled plant was randomly selected and was only sampled if it was standing free of other herbaceous cover and flowering plants. A total of nine plants of each cultivar present were sampled per visit. Vacuum sampling occurred monthly from May to October. Vacuuming of each plant lasted for five seconds. For this work, we assume that sampled wasps were performing foraging or host-seeking behaviors on selected plants (Godfray, 1994). Each sample was frozen until processed by extracting all PH and identifying them to the lowest taxonomic level possible per previous literature (Bennett & Gratton, 2012; Burks & Philpott, 2017; M. H. Egerer et al., 2017). PH identification was accomplished using Hymenoptera of the World (Goulet et al., 1993). Chalcidoidea were identified with the Annotated keys to the Genera of Nearctic Chalcidoidea (Hymenoptera) (Gibson et al., 1997), and Braconidae using the Manual of the New World Genera of the Family Braconidae (Dangerfield et al., 2017). Collected specimens that were damaged were identified to the closest identifiable morphospecies. Cabbage aphids, *Brevicoryne brassicae* were visually identified and abundance was assessed by doing a total count on three random leaves on nine brassicas per cultivar, including counts of apterous, alate, and parasitized aphids. Aphid abundance counts were performed monthly from May to October on non-vacuum sampling days to reduce PH disturbance. Parasitism rates were calculated as number of parasitized aphids divided by number of total aphids on each leaf.

2.3 Data analysis

Generalized linear mixed models (GLMM) were constructed using the MASS R package (Venables et al., 2002) to explore whether APM practices or landscape factors affected PH abundance on common brassicas. Each response variable: All PH, PH superfamily, family, and subfamily abundance, overall site PH diversity, and rates of aphid parasitism were modeled with both local and landscape factors. Local factors include the percent of mulch ground cover, floral and crop richness, production, and non-crop area. Landscape factors include percent impervious surface at 200, 500, and 1000m radii, and farm size. Seasonal factors included both year and season and were assessed as categorical variables: early-season (May to June), mid-season (July-August), and late-season (September-October). The fitdistrplus package in R was used to find appropriate distributions for modeling (Delignette-Muller & Dutang, 2015). A negative binomial or Poisson distribution with a log link function was selected as appropriate given the zero-inflation of the count data. Models were fitted with the glmer.nb or glmer function in R package MASS (Venables et al., 2002). Preliminary models with all measured local and landscape factors were constructed for each response variable. Explanatory variables of low importance for all response variables were excluded from subsequent models. Final models (see Appendix B2) were assessed for fit using the Akaike Information Criterion (AIC) and diagnosed for over or under-dispersion by comparing observed residuals with expected residuals using the DHARMa package in R. Poorly fitted models were excluded from the results (Hartig, 2021). Partial regression plots (predictor effect plots) for final models were developed using the “effects” package in R and are reported in Results (Fox & Weisberg, 2019). The slope of the line in these plots represents the association between a single explanatory variable and a response variable accounting for the effects of each other variable within the fitted model.

3. Results

3.1 Parasitic Hymenoptera sampling

Nine hundred and thirty-eight total vacuum samples yielded 2048 individual PH in the period over 2018-2019. We identified six superfamilies of PH: Ceraphronoidea, Chalcidoidea, Cynipoidea, Ichneumonoidea, Platygastroidea, and Proctotrupoidea, twenty-seven families and fifty-one subfamilies. Our most commonly sampled taxa included the family Braconidae (n=852), and the superfamily Chalcidoidea (n=582), both widely used historically in biological control efforts. The Braconidae included two main subfamilies, Aphidiinae (n=813) and Opiinae (n=39). Sampled families of Chalcidoidea included Pteromalidae (n=224), Aphelinidae (n=136), Eulophidae (n=133), Eucharitidae (n=27), and Encyrtidae (n=19). Four hundred and thirty-three Cynipoidea were collected, including the family Figitidae (n=90), cynipoid subfamily Charipinae (n=59), and the family Eucoilidae (n=47). Three super-families, including two families, and one subfamily were collected in sufficient numbers to be included in the analysis (Table 1).

Table 1. Overview of PH analyzed.

Parasitic Hymenoptera (Data analysis)

Superfamily	Family	Subfamily	*n*=
Chalcidoidea (All)*	-	-	582^{A}
Chalcidoidea	Aphelinidae*	*Unk.*	136
Chalcidoidea	Eulophidae	*Unk.*	133
Chalcidoidea	Eulophidae	Entedoninae	23
Chalcidoidea	Pteromalidae	*Unk.*	224
Cynipoidea (All)*	-	-	464^{A}
Cynipoidea	Figitidae	Charipinae	59
Cynipoidea	Figitidae	*Unk.*	31
Cynipoidea	Eucoilidae	*Unk.*	47
Ichneumonoidea	Braconidae*	-	852^{A}
Ichneumonoidea	Braconidae	Aphidiinae*	813
Ichneumonoidea	Braconidae	Opiinae	39

* = Included in final GLMM models

^{A} = Total number of specimens per superfamily

3.2 Influence of APM practices and local factors on parasitic Hymenoptera abundance and aphid parasitism

Final GLMM models (see Appendix B2, B3) showed significant effects of local and seasonal variables on the sum abundance of several PH taxa at the superfamily, family, and subfamily levels and the parasitism of aphids. No landscape variables had any effect on PH abundance or rates of aphid parasitism.

3.3 All Parasitic Hymenoptera

Models for the abundance of all collected parasitic Hymenoptera showed significant effects of season and local APM factors. The abundance of all PH collected increased with larger non-crop areas on the farm (Fig. 1A). All PH abundance decreased with increased floral richness (Fig. 1B). Despite an increase in collected PH in 2019 (2018, n = 872 and 2019, n = 1007), models that included season as an explanatory variable (early, mid, and late) and year (2018 or 2019) showed a significant overall decrease in PH abundance in late season (Fig. 1C , z = -2.531, P = 0.011).

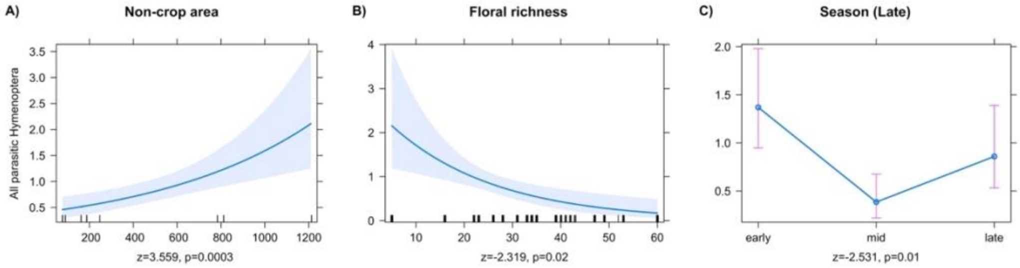


Figure 1. Predictor effect plots for individual explanatory variables on the abundance of all Parasitic Hymenoptera.

3.4 Superfamilies Chalcidoidea and family Aphelinidae

The Chalcidoidea and Aphelinidae showed significant responses in abundance to local explanatory variables. Final models for Chalcidoidea predicted both positive and negative responses in abundance to local factors, including increased abundance with increased crop richness (Fig. 2A), and reduced abundance with increased on-farm floral richness (Fig. 2B). Increased mulch coverage was associated with increased chalcidoid abundance (Fig. 2C). Models for the family Aphelinidae showed significant effects from local variables, including increased abundance with increasing non-crop area (Fig. 2E), crop richness (Fig. 2F), and mulch coverage (Fig. 2G). Chalcidoidea had a mid-season increase (Fig. 2D, z = 4.215, P = <0.001), and late season decrease in abundance (Fig. 2D, z = -3.947, P = <0.001). Aphelinidae abundance increased in both the mid (Fig. 2H, z =2.248, P = <0.024), and late season (Fig. 2H, z =1.904, P =0.056).

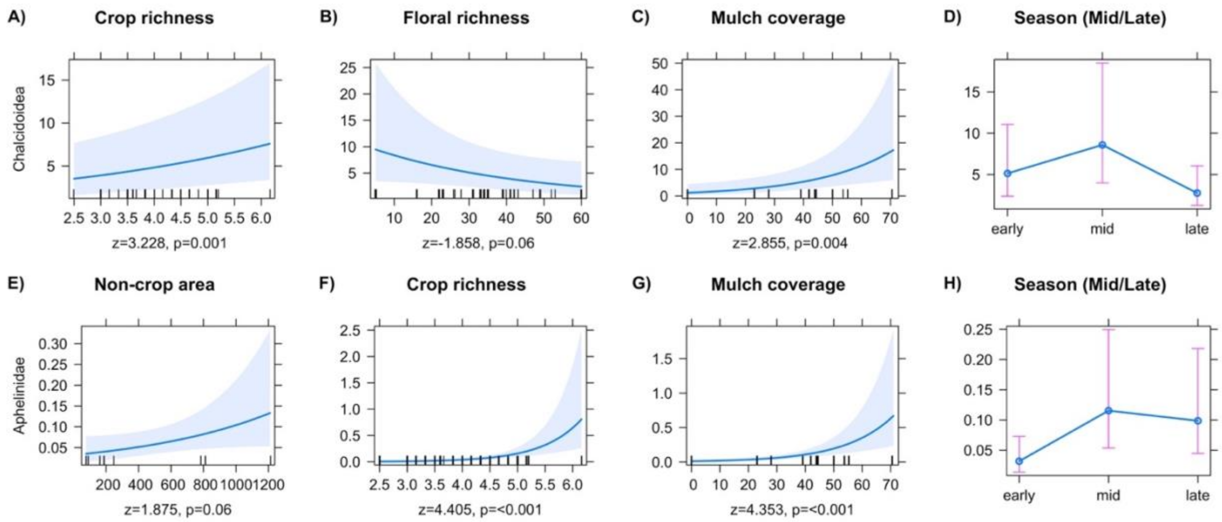


Figure 2. Predictor effect plots for individual explanatory variables on the abundance of superfamily Chalcidoidea (Fig. 2A-D), and family Aphelinidae (Fig. 2E-H).

3.5 Superfamily Cynipoidea

Final models for Cynipoidea showed an increase in abundance with a greater non-crop area (Fig. 3A), and an overall reduction in abundance between 2018 and 2019 sampling periods (Fig. 3B).

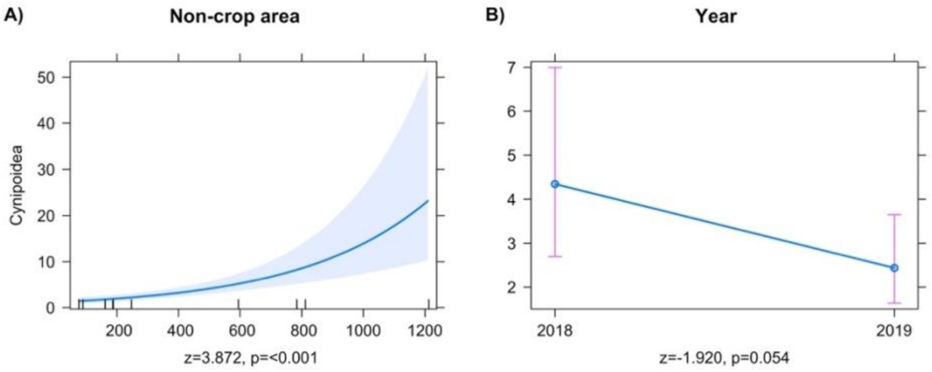


Figure 3. Predictor effect plots for individual explanatory variables on the abundance of superfamily Cynipoidea.

3.6 Family Braconidae and subfamily Aphidiinae

Final models for Braconidae showed a positive response in abundance to increased non-crop area (Fig. 4A). Floral richness reduced braconid abundance (Fig. 4B). Models for aphidiine abundance included local, and temporal explanatory variables in the final model. Increases in the local factors non-crop area increased aphidiine abundance (Fig. 4D). Floral richness reduced aphidiine abundance (Fig. 4E). Aphidiine wasps had a lower abundance in the late season over both sampling years (Fig. 4F, z = -2.841, P = 0.004), but generally had a greater abundance in samples during 2019 (Fig. 4G, z = 2.13, P = 0.033). Across both sampling years, braconid abundance was reduced in both the mid (Fig. 4C, z =-1.971, P = 0.048), and late season (Fig. 4C, z =-4.615, P =<0.001).

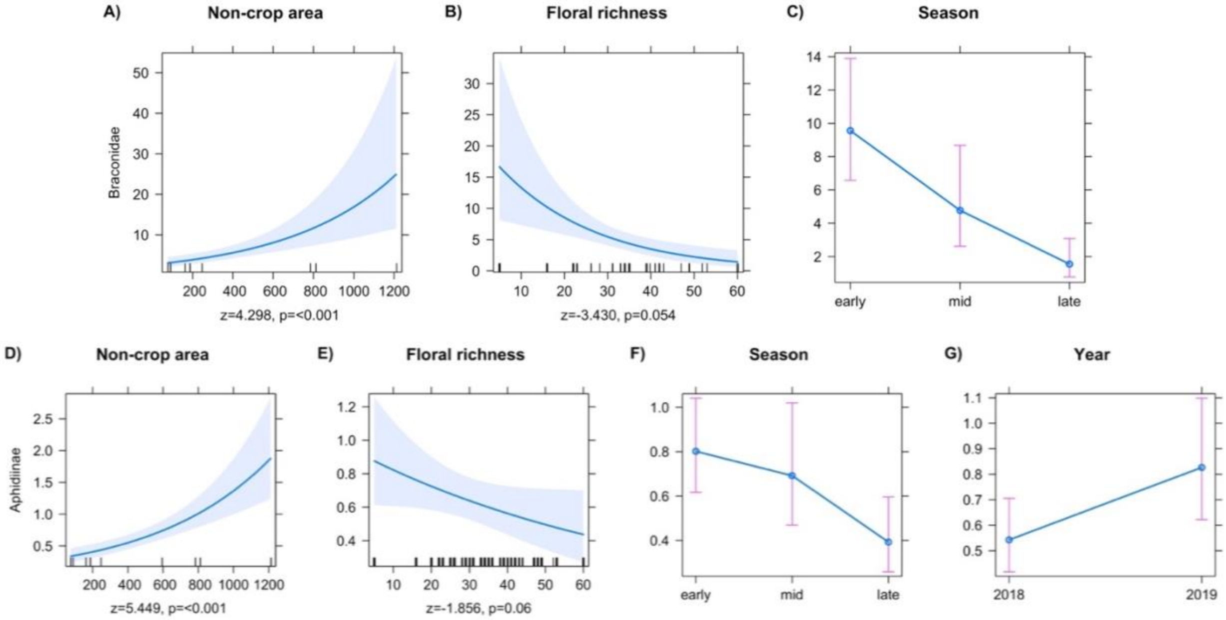


Figure 4. Predictor effect plots for individual explanatory variables on the abundance of Family Braconidae and subfamily Aphidiinae.

3.7 Aphid parasitism

Rates of aphid parasitism increased with crop richness (Fig. 5A). In addition, parasitism rates varied greatly in 2019 with the highest levels measured in mid (Fig. 5B, t = 7.371, P = 0.0001), and late-season 2019 (Fig. 5B, t = 4.897, P = 0.0001).

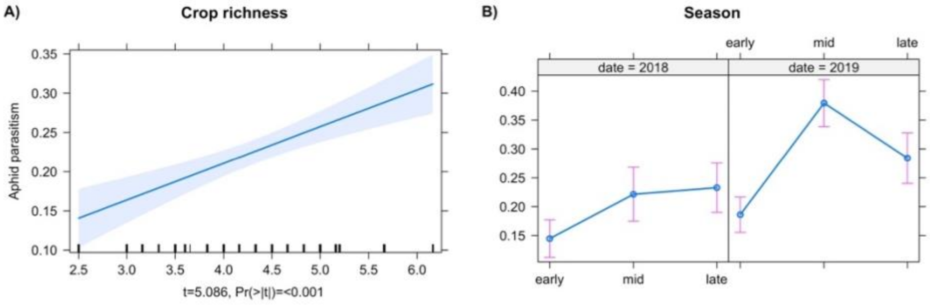


Figure 5. Predictor effect plots for individual explanatory variables (A. Crop richness, B. Season and year) on Rates of parasitism.

4. Discussion

To test the local and landscape effects on the enemies hypothesis vis-a-vis APM on populations of PH in urban agroecosystems, we collected data from twelve urban farms in the San Francisco Bay Area over a period of two growing seasons. Participating farms were selected to represent a continuum of size, spatial composition, and surrounding imperviousness. Non-crop area was a significant predictor for all PH, cynipoid, and braconid wasps. Effects of APM practices were varied, but increased crop richness and mulch coverage were associated with increased abundance of all Chalcidoidea, including the Aphelinidae. Increases in crop richness also showed an increase in parasitism rates of aphids on brassica crop plants. Unexpectedly, Floral richness showed a negative relationship to the abundance of all PH, as well as chalcids, and all Braconidae. All PH showed a significant decline in abundance during the late season of 2019. All measures of impervious surface surrounding urban farms had no effect on PH abundance or aphid parasitism on the urban farms. Landscape effects to arthropod mediated ES continue to have mixed results and this research supports previous findings in urban agriculture which show both negative and positive effects to natural enemy abundance and diversity (Karp et al., 2018).

4.1 On-farm spatial composition

Non-crop areas identified in this research are difficult to identify explicitly as either managed or unmanaged and existed on a spectrum that was often difficult to quantify in interviews or through survey work. However, these areas most frequently had been improved with flowering perennials or annuals, medicinal or “native” flora, and farmers typically stated the purpose as providing a resource for wildlife or beneficial insects. Previous research supports farmer efforts. Structural diversity has been found to elicit positive responses with regard to diversity and abundance of predators and PH in previous UA studies (Burks & Philpott, 2017; M. H. Egerer et al., 2017; Philpott et al., 2014; Tamburini et al., 2020). These areas may provide critical over-wintering habitat in annual cropping systems, additional hosts or prey, shelter, floral nectar resources for nectarivorous insects (Andow, 1991; Landis et al., 2000). Our findings suggest that these non-crop areas have the potential to influence agroecosystem function in UA, and may be an important part of APM practices, even in highly fragmented landscapes. Moreover, floral richness had little effect on PH abundance, or parasitism of aphids, signaling that increase in PH abundance were not due to floral nectar within these non-crop areas. Another mechanism that may be of importance are the spatial composition (or configuration) of the agroecosystem. Our research did not take into account the overall distribution of non-crop area within the farm, which may have failed to account for spatial heterogeneity that has been found to illicit positive and negative biological control responses in agroecosystems (Haan et al., 2020). Future research on urban farms should account not only for the proportion of non-crop areas, but also spatial heterogeneity to further explore these effects.

4.2 APM practices

Overall, APM practices, such as increased mulch coverage and crop plant richness were important predictors of PH abundance, and increased aphid parasitism rates. The connection between mulch, complex groundcovers, and increased abundance and diversity of parasitic wasps has been previously observed in urban agroecosystems (Burks & Philpott, 2017; Morales et al., 2018), a variety of natural habitats, and rural agroecosystems (Langellotto & Denno, 2004). It is unlikely that mulch would provide a direct resource for PH, but PH may benefit from mulch as a potential overwintering habitat or it may provide habitat for potential hosts. Many of the collected PH were parasitoids of dipteran larvae; these larvae are herbivorous but complete part of their life cycle in soils. I suggest that the overall biodiversity of urban farms with increased mulch coverage may create a bottom-up trophic cascade that increases overall soil arthropod diversity benefiting PH populations.

Floral richness had a negative effect on PH abundance in all models. Floral richness was chosen as an explanatory variable as it has previously been found to increase PH abundance in UA (Lowenstein & Minor, 2018). The vast majority of PH are nectarivorous, and this additional nectar resource has been suggested frequently as a strategy for increasing populations, potentially leading to increased parasitism (Andow, 1991; Landis et al., 2000; Langellotto & Denno, 2004; Lee & Heimpel, 2005). However, conflicting data raises questions about this on-farm manipulation and whether PH seek hosts in the same area they feed, or they disperse to increase fecundity (Heimpel, 2019). A large proportion of our overall sample of PH were cynipoids, potentially from the genus *Alloxysta,* known hyperparasitoids of both dominant primary aphid parasitoids in our sample, Aphidiinae, and Aphelinidae (Menke & Evenhuis, 1991). These reductions in primary aphid parasitoid populations may be due to direct or indirect negative effects from this hyperparasitoid that also feeds on floral nectar (Araj & Wratten, 2013; Heimpel, 2019; Heimpel & Jervis, 2005). In urban agroecosystems, floral provisioning as a habitat manipulation may be complicated by the inherent fragmentation and quality of the urban matrix. For floral resources to be an effective APM practice, this resource must be limited. Potential concentrations of alternate off-farm floral resources may complicate this affect.

While this research expanded upon previous findings and can be of utility for urban agroecosystem management, many questions remain. Firstly, the effects of hyperparasitism on biological control in UA. Our third most collected taxon was Cynipoidea, many of which are often hyperparasitoids of aphid parasitizing wasps (Ronquist, 1999). Given that these cynipoids were collected from plant foliage in close proximity to many primary aphid parasitoids, there is some anecdotal evidence that these cynipoids were engaging in host-seeking behavior. If some of the measured on-farm management practices, such as increased non-crop areas also increase abundance of Cynipoidea, this could result in decreased biological control services. In this case, floral provisioning may potentially be acting as an ecosystem disservice (Gillespie & Wratten, 2017; Heimpel, 2019; Heimpel & Jervis, 2005; Zhang et al., 2007). Unfortunately, we were unable to collect parasitized aphids and rear any hyperparasitoids during this research, but these findings suggest that hyperparasitism in fragmented UA landscapes may be a mechanism affecting APM strategies in UA.

Crop plant richness positively affected the abundance of all Chalcidoidea and the subfamily Aphelinidae. Crop richness was also a predictor of greater parasitism rates of cabbage aphids on sampled brassica. Similar findings in rural and urban agroecosystems, including increased PH abundance and biological control services in relation to increased crop diversity have been previously documented (Burks & Philpott, 2017; Isbell et al., 2011; Letourneau et al., 2011; Mace-Hill, 2015; Sperling & Lortie, 2010; Tamburini et al., 2020). Given that intercropping is commonly practiced in UA, these results validate the efficacy of the practice, and offer an opportunity to investigate the extent of the effect in future research efforts.

4.3 Seasonal factors

Seasonal effects on PH abundance were mixed, but many affects were measured in the second year of our sampling. Of note, in 2019, we had fewer sampling events as one farm was unable to participate in our study, but more PH were collected in that year despite the smaller sampling pool. Rates of aphid parasitism were significantly decreased between mid- and late season in 2019. It is unknown what drove these effects, but notable that such a significant difference could occur between sampling seasons. Future research efforts should consider seasonal differences and weather when drawing conclusions about on-farm or landscape factors to PH abundance or diversity or associated biological control services.

5. Conclusions

Our findings support the enemies hypothesis in urban agroecosystems. Local factors, including increased mulch coverage, crop richness, and percent of non-crop areas, were predictors of increased PH abundance and aphid parasitism rates. Our findings support and strengthen previous findings in UA research. Urban farmers should be encouraged to diversify urban agroecosystem spatial composition and implement APM practices to reduce pest impacts.

Chapter 4: Agroecological pest management of aphids via the nectar provision hypothesis in urban agriculture

Abstract

Agroecological pest management seeks to increase biological control services by manipulating on-farm and landscape characteristics in agricultural systems. Common landscape manipulations diversify cropping systems by incorporating natural habitat, inter-cropping, and floral provisioning. Floral resources incorporated into cropping systems can provide nectar and pollen, valuable food resources for parasitic Hymenoptera (PH). Many studies have shown increased parasitoid abundance and richness with increased floral provisioning; they have also demonstrated inconsistent results regarding biological control services. Our study focuses on PH and the nectar provision hypothesis, which states that increased floral provisioning should improve biological control services from PH, reduce pest abundances, and lessen crop damage. We test the nectar provision hypothesis in urban agriculture sites by collecting parasitoids on thirteen flowering plants at eleven urban farms in the San Francisco Bay Area, USA. We collected 664 parasitic Hymenoptera from 6 superfamilies, 25 families, and 14 sub-families. The PH family Pteromalidae were the only PH taxa collected in higher abundance on any observed inflorescence. Parasitic Hymenoptera diversity was higher on marigolds (*Tagetes erecta*) and nettles (*Urtica dioica*). Pest infestations from cabbage aphids (*Brevicoryne brassicae*), crop damage, and rates of aphid parasitism were observed on 3598 brassicas. No increases in aphid parasitism or reduction in crop damage were measured. However, aphid abundance decreased by 10% with every 5% increase in on-farm floral richness. Our research failed to document increased PH abundances or biological control services with increased floral richness but did find an overall reduction in aphid abundance with increased on-farm floral richness.

1. Introduction

Urban and peri-urban agriculture is growing as urbanized areas expand (Faeth et al., 2011; Lawson, 2005; Mok et al., 2014). Urban agriculture (UA) occurs within and near the built environment, with high proportions of surrounding impervious surfaces such as buildings and roads (Lin et al., 2015). Urbanization fragments habitats and reduces the abundance and diversity of organisms (Faeth et al., 2011). Many of the affected organisms are beneficial to urban agriculture and the provision of ecosystem services such as pollination and biological control services (Burkman & Gardiner, 2014; Faeth et al., 2011). Biological control of pest insects is an important ecosystem service for urban farmers due to pesticide use regulation in cities and rejection of chemical management practices for health and environmental reasons. In the absence of chemical controls, agroecological pest management (APM) practices are frequently adopted. Agroecological pest management is a proactive ecosystem services based approach that aims to reduce pest abundance and crop damage by increasing natural enemy populations through agroecosystem diversification (Altieri et al., 2005; Altieri & Nicholls, 2000, 2018; Deguine et al., 2009). However, landscape fragmentation and surrounding imperviousness can often negatively affect the regulatory ecosystem services APM relies upon (Gregory et al., 2016; Oberholtzer et al., 2014). While most off-farm landscape effects are not within the control of urban farmers, on-farm diversification practices are. Substantial research and published literature have investigated the impact of diversification practices to increase biological control of pest insects on rural farms, but less attention has been focused on the effects of diversification in the fragmented landscapes innate to urban agroecosystems (J. E. Arnold et al., 2019; Chaplin-Kramer et al., 2011; Letourneau et al., 2011). This chapter focuses on the effects of floral provisioning on parasitic Hymenoptera (PH) and the ubiquitous cabbage aphid *(Brevicoryne brassicae)*, specifically, the impact of floral provisioning on PH populations vis-a-vis the enemies hypothesis (Root, 1973; Russell, 1989) and the nectar provision hypothesis (Heimpel & Jervis, 2005).

1.1 Enemies hypothesis

Increased plant diversity within agroecosystems, especially intercropping and incorporation of flowering plants, has been shown to increase the abundance of higher trophic level arthropods (natural enemies) that predate on pests, decrease pest abundance, and reduce crop damage (Letourneau et al., 2011; Lichtenberg et al., 2017). Root's enemies hypothesis has been fundamental in understanding these effects. It states that natural enemy populations can be increased through diversification of agroecosystems by offering a variety of resources such as habitat, nectar, pollen, and temporally diverse prey (Landis et al., 2000; Root, 1973; Russell, 1989). However, diversification schemes have not consistently increased biological control services, and Root's enemies hypothesis has been challenged by conflicting findings (Tscharntke et al., 2016).

1.2 Nectar provision hypothesis

The nectar provision hypothesis was proposed to explore the effects of floral-based diversification schemes on the contributions made by PH to biological control services. Heimpel and Jervis posit that with increased accessibility to nectar-producing plants, PH should respond with increased fitness, resulting in elevated levels of localized biological control (Heimpel & Jervis, 2005). Many PH species have been documented feeding on a wide variety of flowers (M. A. Jervis et al., 1993). Nectar, pollen, and extrafloral nectar are essential sources of carbohydrates, proteins, lipids, and minerals for PH (Heimpel & Mills, 2017; M. Jervis & Kidd, 1986; Kehrli & Bacher, 2008; Pemberton & Lee, 1996; Van Driesche & Bellows Jr., 2012). Nectar provisioning has been shown to play an important role in increasing parasitoid longevity, fecundity (Baggen & Gurr, 1998; Kehrli & Bacher, 2008; Patt, 1997), abundance, and diversity (Lee & Heimpel, 2005), as well as increase rates of parasitism (Baggen & Gurr, 1998; Lee et al., 2006; Morandin et al., 2016). Despite documented positive outcomes for parasitoid fitness and increased rates of biological control, floral provisioning does not always result in improved biological control services from PH (Heimpel, 2019; Tscharntke et al., 2016).

In response to confounding results regarding floral manipulations, researchers have proposed several concepts to explain these inconsistencies: 1. PH may utilize floral resources but then disperse to reduce the risk of hyper- or super-parasitism, other mortality, and inbreeding among offspring (Heimpel, 2019); 2. parasitoids may already have enough local floral resources, and floral manipulations may not be introducing a limited resource (Lee et al., 2006); 3. pest insects utilize floral resources more effectively than parasitoids (Baggen & Gurr, 1998; Gillespie

& Wratten, 2017; Winkler et al., 2009); 4. diversification strategies might make it difficult for parasitoids to find hosts in increasingly heterogeneous landscapes (Andow & Prokrym, 1990; Gols et al., 2005); and lastly, 5. many factors determine the ability of PH to use floral resources, including wasp body size, mouthpart morphology, floral structure, and nutritional value (Patt, 1997). A disconnect between plant species and parasitoid feeding characteristics may limit the opportunity of PH to utilize these floral resources (Heimpel & Mills, 2017; Wäckers, 2004). The extent to which these conditions affect PH in urban areas is still being explored.

Generally speaking, the inclusion of flowers into urban agroecosystems to supply nectar for PH should yield effective results in the context of APM. However, inconsistent results regarding the nectar provisioning hypothesis and effects on biological control services have complicated the implementation of floral provisioning practices for farmers. Of all potential remedies for inconsistency regarding effects of floral provisioning, the concept of functional biodiversity has been championed for its potential to influence habitat manipulations that are targeted toward specific ecosystem services or to particular natural enemies. Understanding the linkages between potential PH feeding preferences and specific agroecosystem components could help farmers "fine-tune" their production systems to maximize biological control services. Morphology, bloom time, floral area, and the amount of pollen and nectar resources provided by a given plant species have all been shown to either positively or negatively impact natural enemy populations (Balmer et al., 2014; Fiedler & Landis, 2007; Jado et al., 2019). Gaining a better understanding of the range of flowers most likely to be utilized by and positively affect PH populations and biological control services may enable practitioners to tailor management practices (Baggen & Gurr, 1998; Balmer et al., 2014; Gurr & Wratten, 2000; Jado et al., 2019).

1.3 Pests of concern

Aphids are a common pest in urban agroecosystems. Aphid abundance and resulting crop damage can increase in severity due to various abiotic factors in the built environment. Increased plant resource availability, through irrigation and use of fertilizers, in ornamental landscaping, can make hosts more attractive for herbivorous insects via the plant vigor hypothesis (Cornelissen et al., 2008; M. H. Egerer, Liere, Lin, et al., 2018; Hanks & Denno, 1993; Price, 1991; Raupp et al., 2010). Conversely, poor management of landscaped or ruderal areas, combined with reduced water availability in unirrigated areas and reduced nutrient cycling in urban soils, can stress plants, also making them attractive to pests via the plant stress hypothesis (Galway et al., 1997; Inbar et al., 2001; Kaye et al., 2006; White & McDonnell, 1988; Wortman & Lovell, 2013). Other abiotic factors include the heat island effect (Bowler et al., 2010; Taha, 1997) and fragmentation (De Carvalho Guimarães et al., 2014; Gibb & Hochuli, 2002; Kruess & Tscharntke, 1994). Increased heat can increase abundance by lengthening the duration of the season for multivoltine herbivorous pests (Dale & Frank, 2014; Meineke et al., 2013; Parsons & Frank, 2019). Pest populations can also experience some escape from natural enemies as natural enemy abundance and diversity can be adversely affected by urbanization due to fragmentation and lack of suitable habitat (Bennett & Gratton, 2012; Faeth et al., 2005; Fenoglio et al., 2009; Gardiner et al., 2014; Langellotto & Denno, 2004; Parsons & Frank, 2019). Despite these complicated issues, understanding APM effects in UA may provide urban farmers with a powerful tool to positively impact pest management on their farms.

1.4 Research questions/hypothesis

Crop damage attributed to aphid infestations on brassica crops was a community-driven topic of importance during the development of this research. Our research positioned us as practitioners and researchers. We asked questions specific to refining management options for farmers, including: 1. are there differences in floral resource preference between PH superfamily, family, or subfamily?; 2. do flowers that are commonly planted in urban agriculture sites for pollinators also influence PH abundance or richness?; 3. does increased floral provisioning result in reduced pest abundance or crop damage?; and 4. do floral or crop richness better predict aphid abundance, crop damage, and aphid parasitism?

To answer these questions, we tested the nectar provision hypothesis in participating urban farms. The nectar provision hypothesis states that effective floral provisioning must result in 1. "improved biological control," and 2. nectar-feeding by PH needs to be documented (Heimpel & Jervis, 2005). To test the first criterion, we assessed aphid abundance, parasitism rates for cabbage aphids, and crop damage on commonly grown and culturally relevant Brassicaceae (including kale, collards, broccoli, and tree collards). To test the second aspect of the nectar provision hypothesis, we measured PH abundance and richness on common garden flowers as a proxy for wasp feeding to investigate patterns of feeding preference for potential biological control agents in these urban farms and gardens.

2. Methods

2.1 Sampling methods

In 2018 and 2019, from May to October, eleven community farms in the San Francisco Bay Area participated in insect monitoring with researchers visiting twice a month to sample PH richness and abundance on flowering plants and aphid abundance and crop damage on brassica crops. Participating farms varied in size and landscape composition (see Appendix C) but incorporated similar crop species and management practices.

2.2 Parasitic Hymenoptera sampling

To better understand the effects of floral provisioning on PH richness and abundance and potential feeding preferences, we conducted an in-situ flower survey using an improvised D-vac insect vacuum fitted with a lined and filtered five-gallon bucket that wholly covered flowering plant inflorescences. Each sampled plant was visually assessed for spatial relationships regarding other herbaceous cover and was only sampled if it was standing free of additional herbaceous cover and flowering plants. In addition, each plant was visually assessed for pest infestations and was not sampled if pest infestations were visible. We vacuumed three plants of each flowering plant species present at a farm location. Multiple samples were collected during a farm visits, but varied due to available specimens. Sampling occurred once every thirty days during the same time intervals during each visit. Results from the 2018 survey informed flowering plant selection for the 2019 sampling season. Each plant species that yielded very few or no PH during the 2018 sampling period was excluded from sampling in the following year; 13 flowering plant species were sampled (Table 1). Each sample was stored in a deep freeze until processed by extracting all PH and identifying them to sub-family as per previous literature (Bennett & Gratton, 2012; Burks & Philpott, 2017; M. H. Egerer et al., 2017). Parasitic Hymenoptera identification was accomplished using Goulet et al., (1993) for all groups and Gibson et al. (1997) for Chalcidoidea and Dangerfield et al., (2017) for Braconidae. Collected specimens that were damaged were identified to morphosubfamily.

Table 1. Flowering plant species that were sampled for parasitic Hymenoptera at participating urban farms in 2018-19

Sampled Flowering plants

Common name	Species name
Alyssum	*Lobularia maritima*
Amaranth	*Amaranthus spp.*
Borage	*Borago officinalis*
Calendula	*Calendula officinalis*
Cilantro	*Coriandrum sativum*
Fennel	*Foeniculum vulgare*
Feverfew	*Tanacetum parthenium*
Lavender	*Lavandula spp.*
Marigold	*Tagetes erecta*
Nasturtium	*Tropaeolum (majus or minus)*
Nettles	*Urtica dioica*
Little leaf sage	*Salvia microphylla*
Yarrow	*Achillea millefolium*

2.3 Aphid sampling and plant damage

Aphid abundance, parasitism, and plant damage observations were performed over two growing seasons on commonly grown brassica cultivars: kale, broccoli, collards, and tree collards. Individual plants were randomly selected and identified to cultivar. If possible, we only observed plants that had not been heavily harvested. The major insect pests of interest were cabbage aphids (*Brevicoryne brassicae*), a common agricultural pest of Brassicaceae. Aphid abundance was measured on each plant by selecting three leaves and recording the number of apterous, alate, and parasitized aphids (mummies). The percent of mummified aphids per leaf was used as a measure of biological control services by parasitoid wasps. A qualitative assessment of pest damage on brassicas was completed using a high, medium, and low scale based on familiar concepts of marketability. High damage corresponded to a leaf that would be unmarketable, medium damage had some damage but would still be purchased by a consumer, and low damage had little to no visible damage.

2.4 On-farm characteristics

Two agroecological practices that increase on-farm diversification, floral provisioning, and crop richness (intercropping) were measured three times: early-season (May to June), mid-season (July-August), and late-season (September-October). Crop richness was measured by using 8m transects across cropping systems. Any crop plant that touched the transect line was considered, including different cultivars of the same species (e.g., *Brassica oleracea* cultivars). Three transects were completed on small farms, six on medium farms, and nine on large farms. We collected data for crop richness three times during the growing season over the two years of the study. Floral richness was recorded seasonally, similar to crop richness. Every on-farm, non-crop flowering plant was recorded and identified to genus.

2.5 Data analysis

Generalized linear mixed models (GLMM) were constructed for each of the following response variables: Total parasitic Hymenopteran abundance, super-family, family, and sub-family abundance, and total PH diversity. Selected fixed effect explanatory variables included: floral richness, floral species, year by season, and site as a random intercept. Using the fitdistrplus package in R, parasitic Hymenoptera count data were plotted and examined to determine the best probabilistic distribution for the GLMM modeling; a Poisson distribution with a log link function (Delignette-Muller & Dutang, 2015). Models that had response variables significantly affected by floral species were further analyzed using the non-parametric Kruskal-Wallis test with posthoc Dunn's test to determine floral species that had the most significant influence on the response variable (R Development Core Team, 2010).

Aphid data were analyzed to test for differences in aphid abundance, parasitism rates, and crop damage. Explanatory variables examined were year and season, floral richness?, and crop richness. Aphid count data were assessed using the fitdistrplus package in R to determine the best probabilistic distribution for the GLMM modeling; a negative binomial distribution with a log link function (Delignette-Muller & Dutang, 2015). The final GLMM was constructed with glmer.nb() using crop and floral richness, date and year as fixed effects and site as the random effect. We constructed mixed-effects models using the lme4 package in R (Bates et al., 2015).

After fitting a series of GLMMs based on predictors expected to affect response variables, the model with the lowest Akaike Information Criterion (AIC) score was selected. All GLMM model residuals were simulated from the fitted model using the simulateResiduals function in the package DHARMa to test for dispersion and model fit (Hartig, 2021). Using the effects package in R, a partial regression plot was constructed for each predictor variable included in the final GLMMs (Fox & Weisberg, 2019).

3. Results

In total, 780 floral inflorescence samples were collected in 2018 and 2019, of which 436 contained 664 parasitoid Hymenoptera. The most collected superfamily was Chalcidoidea. Chalcidoid sub-families included were Eulophidae (*n=143*), Pteromalidae (*n=125*), and Eurytomidae (*n=77*). Ichneumonoidea was the second most abundant superfamily (*n= 159*). The family Braconidae wasps accounted for 81% of the total Ichneumonoidea sampled (*n=130*). The two most abundant braconid subfamilies were Aphidiinae (*n=75*) and Opiinae (*n=25*). The third most abundant superfamily collected was the Cynipoidea (*n=37*), followed by the

Platygastroidea (*n=18*). Our final analysis included 518 identifiable parasitoids found on the 13 targeted flowering plant species (Figure 1).

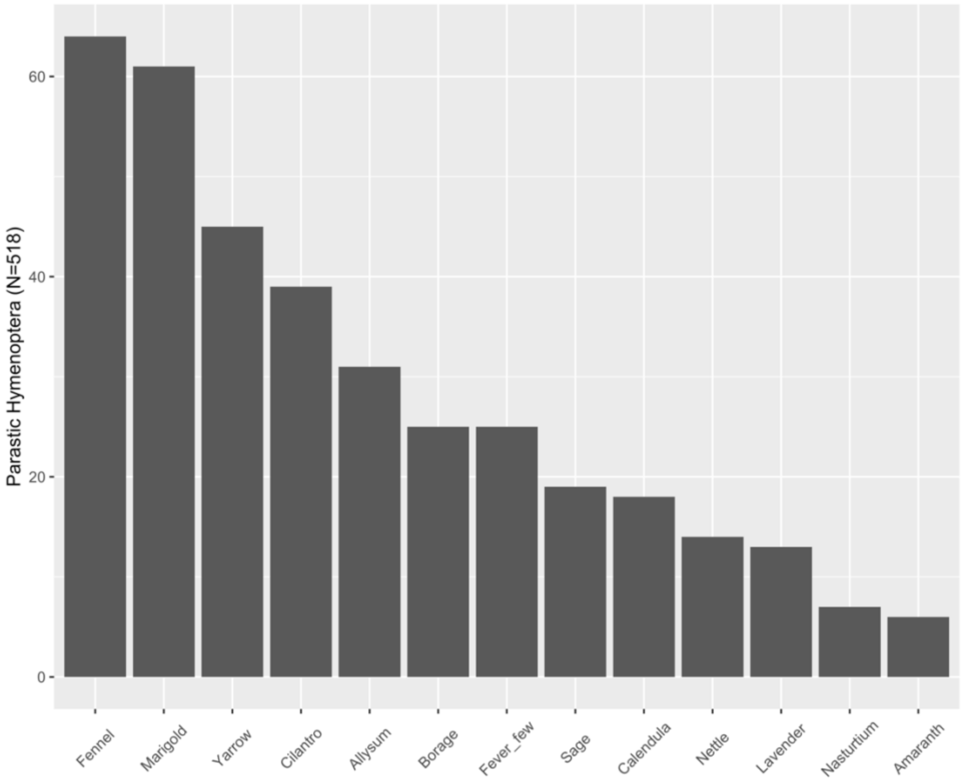


Figure 1. Parasitic Hymenoptera total abundance by flowering plant.

3.1 Parasitoid Hymenoptera abundance and richness

Generalized Linear Mixed Models indicated that floral species was not an explanatory factor for total PH abundance or richness. At the family level, only the abundance of Pteromalidae showed a response to floral richness, specific floral species compared to other parasitoid taxa, and year (Table 2, Figure 2). No sub-families responded to floral richness or showed a feeding preference for floral species. Increased floral richness was a significant predictor of PH richness. However, PH richness model residuals showed high levels of dispersion. To explore the relationship between PH richness and floral species a Kruskal-Wallis test was preformed and a significant difference between PH diversity and floral species (χ^2= 31.391, df = 12, *p* = 0.0017) was measured. Dunn's post-hoc analysis indicated increased PH diversity on marigolds (*p-adj* = 0.043), and nettles (*p-adj* = 0.005) (Figure 3).

Table 2. Generalized linear mixed-modelling results for the parasitic Hymenopteran family Pteromalidae.

Parasitic Hymenoptera modeling (GLMM)

Pteromalidae abundance ~ Floral richness + Floral species + Season*Year + (1|Site), AIC:192

Variable	Est.	SE	*z*-Value	*Pr(>|z|)*
Floral richness	-0.032	0.0102	-3.150	0.002
Floral species (Nettle)	1.081	0.4704	2.298	0.021
Year (2019)	0.529	0.2700	1.960	0.049

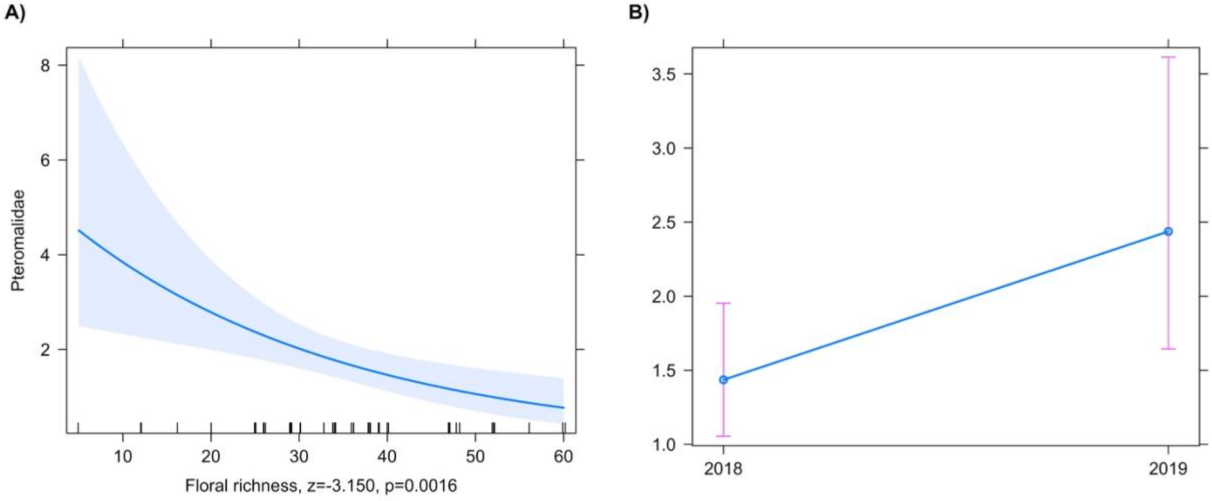


Figure 2. Predictor effect plots for individual explanatory variables on the abundance of family Pteromalidae including floral richness (Fig. 2A), and year (Fig. 2B).

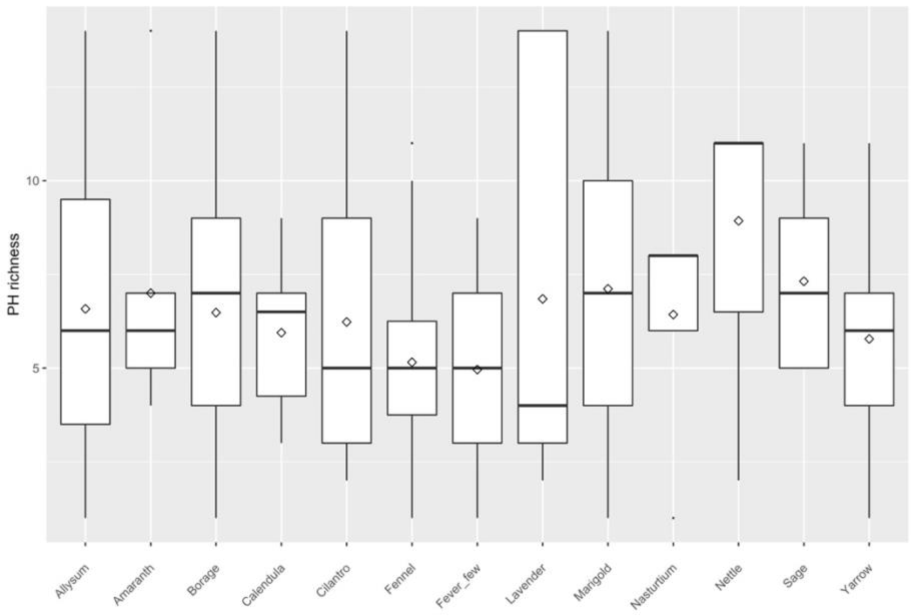


Figure 3. Parasitic Hymenopteran richness and flowering plant species.

3.2 Aphid abundance, parasitism, and crop damage

Three thousand five hundred and ninety-eight brassica were observed for aphid infestations over the two year study period (N=3598). Generalized Linear Mixed Models indicated that aphid abundance decreased with increased floral richness. In addition, in both the mid-season (July-August), and late-season (September-October) of the sampling year 2019 we saw an overall reduction in aphid abundance (Table 3 and Figure 4). Rates of parasitism and crop damage were not found to be affected by any explanatory variables during modeling.

Table 3. Generalized linear mixed-modelling results for cabbage aphid (*Brevicoryne brassicae*).

Aphid modeling (GLMM)

Aphid abundance ~ Floral richness + Crop richness + Season*Year + (1|Site), AIC:17806

Variable	Est.	SE	*z*-Value	*Pr(>|z|)*
Floral richness	-0.015	0.0059	-2.621	0.008
Season (mid), 2019	-1.876	0.221	-8.487	0.001
Season (late), 2019	-1.915	0.2073	-9.238	0.001

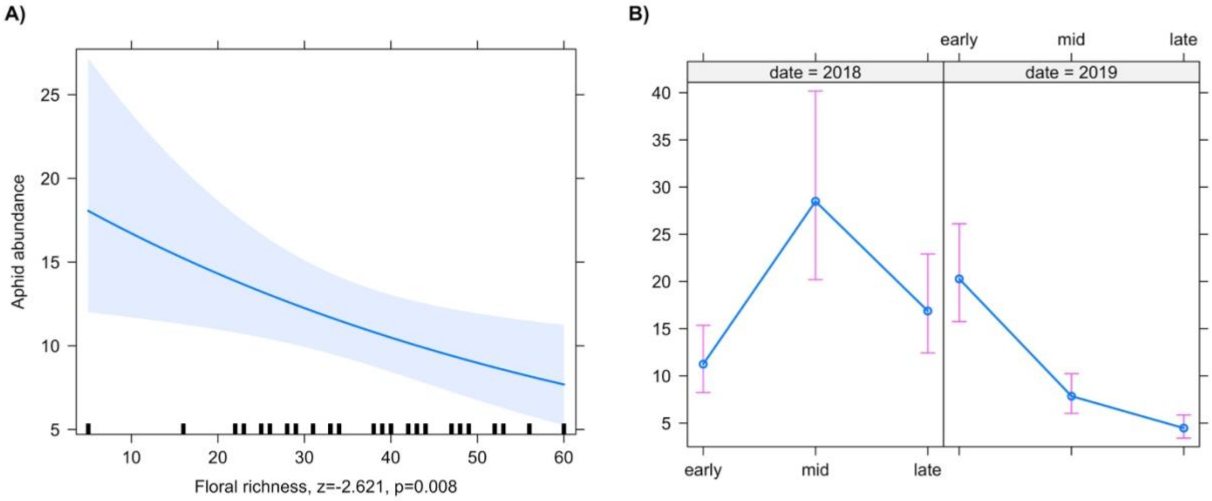


Figure 4. Predictor effect plots for individual explanatory variables on the abundance of the cabbage aphid (*Brevicoryne brassicae*) including floral richness (Fig. 4A), and season (Fig. 4B).

4. Discussion

To better understand the nectar provision hypothesis and the effectiveness of floral-based habitat manipulations, we used PH abundance and richness on flowering plants to indicate parasitoid feeding preferences. Our results showed that floral species were not a strong indicator of increased abundance or richness at any measured scale of PH. Only one PH family, the pteromalids, was found in more significant quantities on nettles. Using a non-parametric test, PH diversity was shown to increase on marigolds and nettles, a response documented by previous floral provisioning research (Andow, 1991; Landis et al., 2000).

Laboratory experiments have shown floral feeding preferences in parasitoids (Baggen & Gurr, 1998; Kehrli & Bacher, 2008), but in-situ results have been less clear (M. A. Jervis et al., 1993). Our results show that our collected PH had a very weak response to floral species, and in some cases showed a negative relationship to floral richness. Several factors may singularly, or in aggregate, explain this absence of floral preference in situ: 1. floral resources incorporated into urban gardens and farms are not selected for their functional diversity but that of other traits, such as attractiveness and availability (Wäckers, 2004); 2. in-situ food resources may present a greater variety of acceptable foods unlike no-choice feeding trials; and 3. some sampling bias may have occurred when using the vacuum on inflorescences as the vacuum may be more likely to capture smaller parasitoids which may be able to exploit a broader range of nectaries or may be feeding on other food items such as honeydew (M. A. Jervis et al., 1993; Kehrli & Bacher, 2008). Many parasitoids also feed on the same plant as their host, which may bias visitation by family and sub-family. A negative PH abundance response to increased floral richness may be a result of dispersion to reduce the risk of hyper- or super-parasitism, other mortality, and inbreeding among offspring (Heimpel, 2019). Our research showed a weak relationship between increased PH richness on marigolds (*T. erecta*) and nettles (*U. dioica*) and an increased abundance response with pteromalids on nettles compared to other PH taxa. Marigolds have a history of being utilized as a beneficial flower in the gardening community and are grown for cultural and aesthetic reasons. Nettles are not typically grown intentionally, and in the few location's nettles were sampled, they were unintentional but preserved in non-crop areas. Nettles, in this case, may be an example of a non-selected floral species with a higher level of ecosystem function than species selected for other traits. Additionally, aphid parasitoids, specifically Aphidius, have been found in higher abundance on nettles due to the occurrence of the stinging nettle aphid, *Microlophiurn carnosum* (Perrin, 1975). It is unclear what connection pteromalids may have in this ecology. It is possible that nettles sampled had infestations of aphids that remained obscured due to the obstacles associated with close inspection of stinging nettles. Despite these findings, anecdotal relationships between floral species and specific PH species indicate these relationships should continue to be explored to better understand parasitoid feeding preferences, floral occupancy, and farmscape mediated biological control.

To assess the second criterion of the nectar provision hypothesis, a demonstratable reduction in pest impacts, we looked at aphid abundance, rates of parasitism, and overall crop damage on brassicas. Our results show that farms with increased floral richness have lower aphid counts per plant. We did not record a reduction in crop damage nor an increase in aphid parasitism with increased floral richness. Our second most abundant category was Braconidae which was dominated by Aphidiinae, parasitoids of aphids commonly found on crop plants at our research sites.

5. Conclusion

Overall, our findings failed to establish the two criteria needed to support the nectar provision hypothesis. Firstly, we did document a reduction in aphids with increased floral richness, but not an increased level of parasitism. A reduction in aphid abundance may be an indirect negative effect due to positive effects of increased floral provisioning on PH, however, we did not document parasitoid feeding preferences on flowers or increased PH abundance with increased floral richness. Some of our results did indicate a PH abundance relationship with floral species (i.e., marigolds and nettles), but these relationships would only be measures of variance between other PH explanatory variables (PH super and subfamily abundance). In summary, the lack of sub-family and family relationships to explanatory variables is not enough to suggest a feeding preference or connect directly to a pest and natural enemy relationship.

Our results do not directly translate into managing elements of functional biodiversity for urban farmers. However, a reduction in aphid populations with increased floral richness is an important finding. Previous research in urban agriculture has similarly found a reduction in aphid populations with floral abundance (M. H. Egerer et al., 2017) and rural agroecosystems (Letourneau et al., 2011). While we did not establish PH feeding preference, recording primary and secondary aphid parasitoids at such high abundance on flowers within urban farms and demonstrating an overall reduction of aphid abundance with increased floral richness suggests that floral provisioning should remain a valued diversification strategy.

Conclusion

Urban agriculture will continue to be an important land use as cities continue to expand and urban populations grow. Outside of the local benefits of food security and sovereignty, growing food in the city provides a myriad of ecosystem services that positively impact environmental conditions, for residents, flora, and fauna. How urban agriculture is managed and incorporated will be a determining factor in how urban agroecosystems can provision ecosystem services. Pest management practices will be an important aspect of managing these systems in a way that promotes the well-being and health of urban farmers and surrounding communities. Agroecological pest management can be an appropriate application for managing pests in UA. Understanding the effects of surrounding landscape composition and on-farm landscape management within the context of the built environment will be an important aspect of this management system.

Results regarding research on landscape and on-farm effects to APM in urban agriculture continue to reveal contradictions, but some trends are emerging, and guidance regarding management practices are now within reach of urban farmers. Our goal was to add to the emerging literature on biological control services in UA by addressing three salient questions that were developed during farm manager interviews over a period of two years. Developing a more nuanced understanding of on-farm management practices, and empowering urban farmers to make decisions based on research performed in urban agroecosystems was an intrinsic goal for this research. Our findings expand on previous findings and suggest that while there may be measurable landscape effects to urban arthropod populations, and their resulting crop damage or biological control services, on-farm management practices continue to emerge as important predictors of ecosystem function in these complex ecosystems. Our findings are presented below within the context of the questions introduced in the introduction:

How do urban farms ecologically function?

We found that adoption of agroecological farming practices is widely practiced across our research sites. Practices such as intercropping, cover-cropping, floral provisioning, and soil building practices are common and often practiced simultaneously. These practices were often cited during interviews with farm managers as strategies to respond to pest pressures, build soil, and produce ecosystem services.

Urban farm production is an important aspect of pest management and biological control, as the efficacy of these systems is often questioned in the context of a production paradigm. Our research showed estimated yields per unit area are high, with approximately 7.14 kg/square meter of fresh vegetables being grown. The most productive farms had in-ground production systems, but in-ground production systems were also associated with increased weed occurrence. Urban farms also significantly impact local food security, with approximately 69% of on-farm production going to the local community.

Despite high yields per unit of area, our on-farm spatial analysis found that an average of only 32% of available area is being utilized for production within the measured urban farms. With land being such a limiting factor of UA adoption, we found that UA sites may not be maximizing potential production area. Previous findings suggest there may be an economic disincentive to increasing production.

Does urbanization perturb ecosystem function and impact biological control services on urban farms?

We completed a systematic literature review to assess the current research of on- and off-farm effects to natural enemies, crop pests, and biological control services. Our review revealed that this topic is understudied but increasing in interest.

Our review identified several consistent on-farm management practices that can positively impact natural enemy populations and resulting biological control services. Increased plant species richness, floral provisioning (increased abundance), increased incorporation of perennials, and increased complex ground cover (mulch), all positively impacted natural enemy populations and resulted in increased biological control services.

Our review found that landscape effects were mixed and inconsistent with all taxa, both positively and negatively affecting natural enemy and pest populations. However, low imperviousness surrounding urban farms had only positive impacts on measured taxa. Our findings reported in chapters 3&4 indicate that all measures of off-farm imperviousness were poor predictors of PH abundance, richness, and parasitism rates. We interpret these findings as an indicator that adoption of specific on-farm management practices positively impact urban agroecosystem function, especially in areas of high imperviousness.

Can farmers overcome impacts of urbanization through on-farm management practices?

All explanatory variables of significance found during modeling were on-farm agroecological management practices. Local factors, including increased mulch coverage, crop richness, and percent of non-crop areas, were predictors of increased PH abundance and aphid parasitism rates. These findings support the enemy's hypothesis in UA.

When assessing impacts of floral provisioning on PH, and resulting biological control services, we found that farms with increased floral richness have lower aphid counts per plant. We did not record a reduction in crop damage nor an increase in aphid parasitism with increased floral richness. We also failed to establish a link between PH and feeding preferences. These findings are consistent with irregular findings regarding biological control services, PH, and floral provisioning set forth in Chapter 4. However, our literature review reveals that floral abundance does have frequent beneficial effects to PH and resulting biological control services. With the support of the enemies hypothesis established in Chapter 3, the reduction of aphid populations in Chapter 4, and the previous studies reported in the literature review, we believe there is strong evidence to support that urban farmers can significantly, and positively, impact urban agroecosystem function through management practices.

Findings and questions

This research supports the enemies hypothesis in UA and empowers urban farmers to adopt on-farm management practices that increase agroecosystem function and increase ecosystem services. However, we did find conflicting results, floral provisioning is an important predictor of increased abundance of natural enemies, including PH, and increased biological control services in many urban specific and rural farm diversification research. Our findings indicated that floral richness was a strong predictor, across many PH taxa, of reduced abundance. Two factors presented by Heimpel and Jervis in 2009 may be important factors in this observed reduction; 1. In areas of high floral occurrence, parasitoids may be seeking hosts away from concentrations of food sources as these areas may increase the opportunity for hyper-parasitism. Hyperparastioids, specifically the cynipds, were a large proportion of our collected wasps. Many of these cynipoids, specifically the subfamily Alloxystinae are known secondary parasitoids of baraconid aphid parasitoids. In both chapters 3&4, these PH were found in high abundance, indicating that there may be significant pressure from secondary parasitoids in areas of high floral nectar. Secondly, floral nectar may not be in great demand in urban areas. Landscaping with flowering plants is common and may be introducing an important confounding variable. Given these findings, floral provisioning (or at least increased floral richness) may not be promoting biological control services in urban farms.

A second salient explanatory variable that had a positive impact on PH abundance across measured farms was on-farm non-crop areas. Many previous research efforts regarding ecosystem function in fragmented landscapes have focused on matrix and patch quality as explanatory variables for species occurrence or resulting ecosystem services in fragmented landscapes. While these factors continue to be compelling, recent meta-analysis call into question on-farm spatial composition as an important determinant of ecosystem function (Haan et al., 2020). These non-crop areas differed greatly during sampling, but were often distributed throughout the farm. Future research on agroecosystem function should pay closer attention to the composition and evenness of these often overlooked ruderal patches within urban farms.

A note on other important natural enemy taxa: Our research explicitly looked at aphids and parasitic Hymenoptera in urban agroecosystems. Both PH and aphids have been shown to be resilient in the context of urbanization and fragmentation in agroecosystems. Less mobile taxa, such as ground beetles and spiders have shown significant declines with habitat fragmentation. Overall ecological function of cities should be an important topic in future discussions regarding urban planning and growth. Important ecosystem services, linked with declines in diversity of terrestrial invertebrates should not be overlooked in the context of this research.

Implications of the research

Biological control services will continue to be a salient topic in urban agroecosystem management. Our findings indicate that on-farm management practices, including spatial composition and land use within urban farms, can be important predictors of biological control services and should be considered during agroecological pest management plans. On-farm agroecosystem diversification, including increased crop biodiversity, incorporation of complex ground covers, and on-farm habitat refuges can reduce pest abundance and increase natural enemy abundance in urban agroecosystems.

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Appendix A

(A1) Survey questions

Survey questions.

1. Name of garden/farm
2. Date
3. Interviewee name
4. Relationship with farm?
5. When was the farm established and by who?
6. What is the mission of the farm?
7. Can you offer a brief history of this farm?
8. What are the economic goals of the farm?
9. What is the relationship with the legal landowner?
10. Do you pay any form of rent?
11. Is there a limit on your time here?
12. Is the farm affiliated with any particular institution? (e.g., religious organization, housing site, school, non-profit, network of community gardens)
13. What type of support does the institution provide? (Financial, agronomic, political etc.)
14. How many people farm/work here?
15. What is their relationship to the farm?
16. How do people learn about and get involved in this particular farm?
17. Can community members access the farm at any time?
18. Does the farm have something like "open-hours" or supervised farming periods each week?
19. How does the community interact with this farm? (e.g. people from the community farm here, neighbors visit the farm, etc.)
20. What impact do you think this farm has had on the surrounding area/neighborhood?
21. How did the farm acquire tools/implements?
22. Where do you get your seeds/seedlings from?
23. Is the farm irrigated?
24. Type of irrigation?
25. How much water is used?
26. Where does the water come from?
27. Do you pay for the water?
28. Is there infrastructure for water harvesting?
29. Are you facing any challenges due to the drought? How are you overcoming those?
30. What do you produce? (Annual/Perennial crops; Flowers/ornamentals; Animals (beehives, chickens, goats, pigs etc.) and for what use (honey, wax, meat, milk, eggs, hide, are they just pets etc.)
31. What are the main pest problems (by crop family or species)?
32. Methods to prevent pests?
33. Methods to deal with pest infestation/outbreak?
34. What do you think is the effectiveness of these methods? What has worked well or not?
35. Does the farm have native or natural habitat areas?
36. What are the main pathogenic species? Foliar, soil-borne etc.?
37. Methods used to prevent diseases?

38. Methods used to deal with disease following infestation/outbreak?
39. What do you think is the effectiveness of these methods? What has worked well or not?
40. What are the main weed species? Annual vs. perennial; grasses vs. broadleaves?
41. What do you do to prevent weeds? (e.g. reduce weed seedbank, prevent seedling germination, avoid weed seed production, etc.)
42. What do you think is the effectiveness of these methods? What has worked well or not?
43. How do you manage soil quality/fertility?
44. Do you use any of these practices? Recycle biomass; add soil nutrients; add organic matter (mulch and/or compost?)
45. Where do you get amendments?
46. How do you prevent (minimize) nutrient loss?
47. Do you use cover crops?
48. What practices do you use to conserve water and soil moisture?
49. Do you produce compost on-site?
50. Has the soil at this site been tested?
51. Do you know what it was tested for?
52. How did you decontaminate soil or avoid soil contaminants?
53. What inputs come from off farm?
54. What do farmers do with their harvest? (e.g., personal consumption, donated, trade/barter, sold in the market, etc?)

(A2) Farm participation in research

Farm participation (N = 29)							
Survey	Production area	Size	Impervious surface (%)	Yield	Weeds	Ground cover	Crop biodiversity
25	25	25	12	17	15	13	13
86%	86%	86%	41%	58%	51%	44%	44%

(A3) Description of on-farm management practices

Inter-cropping – Growing of two or more crops simultaneously in close proximity

Mulch – Using organic matter, often fallen leaves or wood chips, to cover bare soil in production areas

Crop rotations – Spatially shifting cultivation of particular crop families seasonally

Green manure – Incorporating green crop residues into the soil

Double-digging – Method of incorporating organic matter into garden beds through hand-tillage. The method is typically attributed to the French-intensive style of gardening.

No-till – Soil management practices that reduce disturbance of soils. This technique is regionally associated with soil management practices similar to those implemented at Singing Frog Farms in Sebastopol, California

(A4) Descriptive Statistics, Kruskal-Wallis Test (Weed %/m^2)

	H	DF	*p-value*	Z	n	Mean	sd	Min	Max	% zero
	30.434	1	3.45E-08	-5.516696	-	-	-	-	-	-
Raised bed production	-	-	-	-	372	4.47	6.95	0	40	0.48
In-ground production	-	-	-	-	560	8.36	12.79	0	100	0.34
	7.1671	1	7.43E-03	2.677151	-	-	-	-	-	-
Not intercropped	-	-	-	-	97	13.350516	20.26802	0	100	0.38
Intercropped	-	-	-	-	835	6.053892	9.080811	0	60	0.4

Appendix B

(B1) Farm spatial characteristics and location

Site coordinates (Plus Code)	Farm size (m^2)	Production area (m^2)	Non-crop area (m^2)	Impervious surface % (200m radii)	Impervious surface % (500m radii)	Impervious surface % (1km radii)
VMJX+P8 Berkeley, California	8903	1028	784	69	67	67
VPCR+QH Berkeley, California	5712	2300	0	75	67	62
RPC7+VJ Oakland, California	5308	381	1213	84	78	80
WMWM+4F Richmond, California	4477	966	595	56	49	41
QR5P+PQ Oakland, California	3892	1998	162	62	61	57
VP87+2J Berkeley, California	2348	760	188	66	60	61
VPH6+FF Berkeley, California	2299	1007	277	55	57	58
XM23+R6 Richmond, California	1428	206	77	34	57	64
QR27+MX Oakland, California	968	111	247	64	71	75
QPVQ+V8 Oakland, California	932	140	52	52	64	74
VP39+46 Berkeley, California	799	178	247	63	62	64

(B2) Final models	AIC
All parasitic Hymenoptera ~ Date + Season + Average mulch coverage + Crop richness + Floral richness + Non-crop area + Site*	1425
All Chalcidoidea ~ Date + Season+ Average mulch coverage + Crop richness + Floral richness + Site*	529
Chalcidoidea Aphelinidae ~ Date + Season + Average mulch coverage + Crop richness + Total size + Production size + Site*	331
All Cynipoidea ~ Date + Season + Crop richness + Non-crop area + Site*	407
Ichneumonoidea Braconidae ~ Date + Season+ Average mulch coverage + Crop richness + Floral richness + Site*	332
Ichneumonoidea Braconidae Aphidiinae ~ Date + Season + Floral richness + Non-crop area + Site*	1768
Rate of parasitism ~ Date*season + Crop richness^A	-143

*=Site as random effect to control for psuedoreplication

^{A} = Modelled as GLM as the variable "site" had no effect

(B3) GLMM Results

Response variable	Explanatory variable	Est.	Std. err	z-Value	Pr(>|z|)
All parasitic Hymenoptera	Area of non-crop	0.0013	0.0004	3.559	0.000372
-	Floral richness	-0.031483	0.0136	-2.319	0.020402
-	Season (Late)	-0.828814	0.3274	-2.531	0.011363
Chalcidoidea	Crop richness	0.2099	0.065	3.228	0.00125
-	Floral richness	-0.0247	0.0133	-1.858	0.06311
-	Mulch coverage	0.0377	0.0132	2.855	0.0043
-	Season (Mid)	0.5145	0.1221	4.215	0.000025
-	Season (Late)	-0.62076	0.1573	-3.947	0.000079
Aphelinidae	Area of non-crop	0.0012	0.0006	1.875	0.0608
-	Crop richness	1.4113	0.3204	4.405	0.000011
-	Mulch coverage	0.0589	0.0135	4.353	0.000013
-	Season (Mid)	1.29	0.5738	2.248	0.0246
-	Season (Late)	1.1325	0.5947	1.904	0.0569
Cynipoidea	Area of non-crop	0.0023	0.0006	3.872	0.000108
-	Year (2019)	-0.650749	0.3389	-1.92	0.05481
Braconidae	Area of non-crop	0.0018	0.0004	4.298	0.000017
-	Floral richness	-0.044612	0.013	-3.43	0.000603
-	Season (Mid)	-0.695516	0.353	-1.971	0.048778
-	Season (Late)	-1.820861	0.3945	-4.615	0.000004
Aphidiinae	Area of non-crop	0.0015	0.0003	5.449	5.08E-08
-	Floral richness	-0.012649	0.0068	-1.856	0.06345
-	Season (Late)	-0.714486	0.2515	-2.841	0.0045
-	Year (2019)	0.4209	0.1976	2.13	0.0332
		Est.	Std. err	t-Value	Pr (>|t|)
Aphid parasitism [A]	Crop richness	0.0468	0.0092	5.086	4.59E-07
-	Date (2019):season (Mid)	0.1419	0.0193	7.371	4.34E-13
-	Date (2019):season (Late)	0.0942	0.0192	4.897	4.34E-13

[A]= When modelled in GLMM, the random effect (Site) were not present. Therefore, modelling for rates of parasitism were completed with GLM

Appendix C

(C) Farm characteristics

Site coordinates (Plus Code)	Farm size (m^2)	Production area (m^2)	Non-crop area (m^2)
VMJX+P8 Berkeley, California	8903	1028	784
VPCR+QH Berkeley, California	5712	2300	0
RPC7+VJ Oakland, California	5308	381	1213
WMWM+4F Richmond, California	4477	966	595
QR5P+PQ Oakland, California	3892	1998	162
VP87+2J Berkeley, California	2348	760	188
VPH6+FF Berkeley, California	2299	1007	277
XM23+R6 Richmond, California	1428	206	77
QR27+MX Oakland, California	968	111	247
QPVQ+V8 Oakland, California	932	140	52
VP39+46 Berkeley, California	799	178	247