Title

Local and Landscape Drivers of Biodiversity, Ecosystem Services, and Human Well-being in Urban Agroecosystems

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Publication Date

2019

UNIVERSITY OF CALIFORNIA
SANTA CRUZ

LOCAL AND LANDSCAPE DRIVERS OF BIODIVERSITY, ECOSYSTEM SERVICES, AND HUMAN WELL-BEING IN URBAN AGROECOSYSTEMS

A dissertation submitted in partial
satisfaction of the requirements for
the degree of

DOCTOR OF PHILOSOPHY

in

ENVIRONMENTAL STUDIES

by

Monika H. Egerer

June 2019

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Abstract

Local and Landscape Drivers of Biodiversity, Ecosystem Services, and Human Well-being in Urban Agroecosystems

Monika H. Egerer

This dissertation examines the local and landscape drivers of biodiversity, ecosystem service provisioning and human well-being in urban agricultural systems (agroecosystems). I explore three major themes: 1) urban biodiversity and ecosystem services; 2) climate change, water management and sustainability; and 3) urban gardens as socioecological systems. In an interdisciplinary approach, I use quantitative and qualitative methods in natural and social systems.

In the first section, I examine the local and landscape scale drivers of biodiversity and pest control within urban gardens. I focus on the abundance and species richness of ladybird beetles – an important and mobile natural enemy of garden pests. I measure ladybeetle diversity within gardens, and experimentally test factors predicted to influence ladybeetle fidelity to gardens. I find that ladybeetle dispersal is higher from gardens in more impervious landscapes, albeit overall high taxonomic richness supported by these habitats. In comparison to other contexts, these results may be unique to California due to water availability maintained through garden irrigation.

In the second section, I investigate how gardens become irrigated oases in an otherwise drought landscape. I use citizen science to collect water use data and ask how environmental concern and water governance are affecting water use behavior by gardeners. I also investigate how climate variability and increasing extremes influence water management and plant species cultivated in gardens. I find that gardeners lack an understanding of water use, are responding to weather patterns by changing watering behaviour, and that garden rules can limit water use. Moreover, research participation can improve gardeners' water conservation literacy. This work informs our understanding of how climate change may impact urban agriculture sustainability and alludes to the socioecological complexity of gardens.

In the third section, I focus on the social aspects of urban gardens and their management. I show that gardens provide many well-being benefits to gardeners, and are used more by people in urban areas. Yet I explain that there are crucial issues that undermine social and ecological sustainability in these gardens: the resources that create a habitat for biodiversity dually create social-political rifts within gardener communities. Gardens, and their biophysical and social elements are relatively novel in the scope of contemporary land use transformations.

In summary, I show that urban gardens are sites of biodiversity, climate challenges, and social-ecological complexity that add to urban novelty. Using our understanding of complexity can inform management to improve urban and agricultural sustainability.

Acknowledgments

This work would not have been possible without the support of various faculty, mentors, undergraduate students, graduate students, community members, funding agencies, family and friends. To all of them, I am eternally grateful.

First, I sincerely thank the field and lab work provided by dozens of undergraduate and graduate students that contributed to this work: C. Adams, Y. Bravo, J. Burks, Y. Byun, M. Cely-Santos, H. Cohen, E. Connor, V. De la Fuente, K. Ennis, K. Forbush, A. García, P. Gillette, M. Hash, D. Hafalia-Yackel, R. Hruska, B. Ferguson, E. Jimenez-Soto, Z. Jordan, C. Kirk, M. MacDonald, L. Marin, T. Milz, H. Morales, J. Muramoto, M.E. Narvaez Cuellar, T. Ong, A. Ossola, M. Otoshi, M. Plascencia, R. Quistberg, A. Rubio, C. Sanchez, G. Santiz Ruiz, R. Schreiber-Brown, J. Shaw, H-A. Tan, J. Tan, S-S. Thomas and M. Zavatta. Thank you to the UCSC Greenhouse Staff M. Dillingham and J. Velzy for growing experimental plants used in the research. Thank you to C. Anton, D. Bonett, A. Singh and J. Suraci for statistical assistance. The data and analyses supporting this dissertation would not have been possible without these various people.

I thank the community gardens and gardeners for supporting and facilitating my research over five years. Further, I thank the gardeners for sharing their stories and often the fruits of their labor with me. I have gained a tremendous insight about agroecology, human values, and social-ecological complexity through my various interactions with them. I thank gardens in the California Central Coast including: Aptos Community Garden, Beach Flats Community Garden, Berryessa Community

Garden, the Center for Agroecology and Sustainable Food Systems, Charles Street Community Garden, Chinatown Community Garden, Coyote Creek Community Garden, El Jardín at Emma Prusch Park, The Forge at Santa Clara University, Giving Garden at Faith Lutheran Church, Homeless Garden Project, La Colina Community Garden, Laguna Seca Community Garden, Live Oak Green Grange Community Garden, MEarth at Carmel Valley Middle School, Mesa Verde Gardens, Mid-County Senior Center, Our Green Thumb Garden at Middlebury Institute of International Studies, Pacific Grove Community Garden, Salinas Community Garden at St. George's Episcopal Church, and Trescony Community Garden. I thank gardens in Melbourne, Australia, including: Ashburton Community Garden, Balwyn Community Garden, Box Hill Community Garden, Essendon Community Garden, Flemington Community Garden, Hawthorn Riversdale Road Community Garden, Merri Corner Community Garden, Nunawading-Jolimont Community Garden, Nunawading-Slater Community Garden, Rushall Community Garden, and West Brunswick Community Garden and Food Forest.

I graciously thank many significant mentors that have shaped the way that I think and work: Peter Bichier, Madeleine Fairbairn, Evan Fricke, Binney Girdler, Shalene Jha, Dave Kendal, Heidi Liere, Brenda Lin, Alessandro Ossola, Camilo Ordonez-Barona, Stacy Philpott, Haldre Rogers, and Erika Zaveleta. Thank you for your time, support and advising. In particular, I thank Dr. Madeleine Fairbairn for her guidance in qualitative research, from field work to data analysis to writing. Her support through conversation, literature, and written feedback greatly contributed to my intellectual growth in political ecology and sociology. I thank Dr. Brenda Lin for being a role model for how to think and work interdisciplinary. I thank her for teaching me how to work and to collaborate with colleagues across disciplines and across international borders (and time zones). The opportunity she provided to me to be a colleague gave me the confidence to be an independent scientist. I sincerely thank Dr. Stacy Philpott for her mentorship as my primary advisor. Her capacity to do excellent research, provide thoughtful mentorship, and be a very genuine person is inspiring. I thank her for her time, her honest feedback, and for consistently challenging me to improve. I have learned an immeasurable amount from her about research, academia, and fostering an academic community. Thank you, Stacy, for everything that you do and for believing in me as a student and researcher.

Thank you to inspiring lab mates and dear friends in the Philpott research group that have become my family. I especially appreciate and thank: Alicia Calle, Hamutahl Cohen, Marcela Cely-Santos, Estelí Jiménez-Soto, and Elissa Olimpi. Thank you to Estelí Jiménez-Soto for your comradery, and your contagious enthusiasm about ants, burgers, and everything in between. Thank you to Hamutahl Cohen for your collaboration, your honesty, and for teaching me the ropes of graduate school early.

Thank you to the 2014 cohort of graduate students for the companionship, support and comradery. Thank you to Colby Anton, Rick Flores, Emily Reisman, Rachel Voss, Justine Smith, Anna Nisi, Alejandro Artiga-Purcell, and many others for their friendship in and out of graduate school. Special thanks to Colby Anton for

keeping the office not too serious, for conversations on tracking ladybugs and cougars, and for being a good friend. You improved my well-being significantly through laughter and by introducing me to ergonomic desks.

Thank you to my family for their continuous support over the years, especially Manfred Egerer, Liz Hugel, Martina Egerer, Rainer Egerer and Willibald Eck.

Last, I thank the dozens of anonymous reviewers and journal editors for providing critical feedback that challenged me to continuously improve my work.

My research was financially supported by the National Science Foundation Graduate Research Fellowship Program. In addition, financial support was provided through: the National Institute of Food and Agriculture, United States Department of Agriculture, the Center for Agroecology and Sustainable Food Systems, the Environmental Studies Department at the University of California, Santa Cruz, and an Annie's Homegrown Sustainable Agriculture Scholarship. Thank you.

The text of this dissertation includes reprints of the following previously published material:

- Lin BB and Egerer M (2017) Urban agriculture: an opportunity for biodiversity and food provision in urban landscapes, *Urban Biodiversity: From Research to Practice*. Taylor & Francis/Routledge, London, UK.
- Egerer MH, Bichier P, Philpott SM (2016) Landscape and local habitat correlates of lady beetle abundance and species richness in urban agriculture. *Annals of the Entomological Society of America.* 110(1): 97-103.
- Egerer MH, Liere H, Bichier P, and Philpott SM. 2018. Cityscape quality and resource manipulation affect natural enemy biodiversity in and fidelity to urban agroecosystems, *Landscape Ecology*. 33(6): 985-998.
- Egerer M, Liere H, Lin B, Jha S, Bichier P, Philpott SM. (2018) Herbivore regulation in urban agroecosystems: direct and indirect effects, *Basic and Applied Ecology*. 29: 44-54.

- Egerer M, Li K, Ong TYW (2018) Context matters: contrasting ladybird beetle responses to urban environments across two US regions. *Sustainability* 10:1–17.
- Egerer M, Lin BB, Philpott S. Water use behavior, learning and adaptation to future change in urban gardens. *Frontiers in Sustainable Food Systems*. 2(71): 1-14.
- Egerer M, Lin BB, Threlfall CG, Kendal D. (2018) Temperature variability influences urban garden plant richness and gardener water use behavior, but not planting decisions, *Science of The Total Environment*. 646:111-120.
- Egerer M, Philpott SM, Bichier P, Jha S, Liere H, Lin BB. (2018) Gardener well-being along social and biophysical landscape gradients, *Sustainability* 10(96):1–14.
- Egerer M and Fairbairn M. (2018) Gated gardens: Effects of urbanization on community formation and commons management in community gardens, *Geoforum*. 96:61-69.
- Egerer M, Ossola A, Lin BB (2018) Creating socioecological novelty in urban agroecosystems from the ground up. *Bioscience* 68:25–34.

The co-author listed in this publication directed and supervised the research which forms the basis for the dissertation. All co-authors approve the inclusion of this work in the dissertation.

Introduction

Urban agriculture: integrating biodiversity conservation and ecosystem services within agroecosystems in urban landscapes

Adapted from Chapter 5 in the book *Urban Biodiversity: From Research to Practice*, 2018

Abstract

Urban agriculture has recently experienced a renaissance in many cities because of its multifaceted contributions to urban life. In many cities around the world, food production from peri-urban and urban lands provides significant quantities of food to urban dwellers in a sustainable fashion, connecting people to food production and local industries. In many post-industrial cities, urban agriculture has presented itself as an important source of nutritious and healthy food, increasing the access of many city dwellers to fruits, vegetables, medicinal plants, as well as culturally appropriate foods. Urban agriculture, additionally, provides a vital green space within cities capable of supporting biodiversity and ecosystem services. The broad range of urban agricultural systems (community gardens, private gardens, rooftop gardens, and more), as well as the diverse set of crop and non-crop plants supported by these

systems, can provide a complex assemblage of vegetation that allows for both planned and associated biodiversity to persist within the urban matrix. The improved ecosystem services not only support more resilient food production systems by protecting ecosystem functions such as natural pest control and pollination, but also provide important community services such as a sense of place and belonging. However, the biodiversity and ecosystem services on the local level can be displaced if vegetation changes at the landscape level threaten the ability of beneficial insects and animals to persist in urban gardens or negatively impact food production. Thus, the design of urban agriculture, and the surrounding urban landscape, can have important repercussions on biodiversity, food production, and overall associated benefits from these systems. In this introductory chapter, I provide information to preface the motivation for the empirical and experimental studies that I performed for this dissertation

Introduction

In the face of urbanization and growing human populations, urban green spaces play an important role in harboring biodiversity within city landscapes, maintaining trophic interactions and food web stability, and providing ecosystem services to urban residents (Goddard *et al.* 2010). Urban agriculture – the production of food and livestock in urban areas – is an important feature of urban green infrastructure. These agro-ecosystems can be considered islands of high biodiversity in the urban landscape as they generally contain an abundance of species of plants, birds, and arthropods in comparison to the surrounding urban landscape matrix (Goddard *et al.* 2010). This is especially true for beneficial insects like pollinators (e.g., Ahrne *et al.* 2009) and natural enemies (e.g., Bennett and Gratton 2012). Organisms like bees, flies, butterflies, spiders and beetles provide key ecological functions through pollination and pest predation, which lead to ecosystem services that increase plant and crop production in both rural (Losey and Vaughan 2006) and urban agricultural systems (Lin *et al.* 2015). Urban agriculture is thus a key space for biodiversity and ecosystem service provisioning that can increase local food production and urban food security and access (Smit *et al.* 1996). As urban populations grow across the world, urban agriculture is becoming ever-more important for its socio-economic implications like increased food security and nutrition (Alaimo *et al.* 2008), its significant role in biodiversity conservation and in urban ecology (Lin *et al.* 2015), and overall integrating multifunctionality into densely populated urban landscapes (Lovell 2010).

What is Urban Agriculture?

Urban agriculture (UA) is defined as the production of crop and livestock goods within cities and towns (Zezza and Tasciotti 2010), and it is generally integrated into the local urban economic and ecological systems (Mougeot 2010). UA often includes peri-urban agricultural areas around cities and towns, which may provide products and services to the local urban population (Mougeot 2010). Urban agriculture activities are diverse and can include the cultivation of vegetables, medicinal plants, spices, mushrooms, fruit trees and other productive plants, as well as keeping livestock for eggs, milk, meat, wool and other products (Lovell 2010). The different types of UA contribute to the edible landscape in a range of community types and provide a broad array of services based on community needs and desires (McLain *et al.* 2012). This can include spaces such as private gardens (household area privately cultivated), community gardens (areas collectively cultivated), allotment gardens (parcelled areas individually cultivated), and peri-urban farms (production-focused systems) (Table 1). UA systems are highly heterogeneous in size, form and function and can be found in different types of urban green spaces. This diversity is based on some important factors including land tenure, management, production type, and scale of production.

Many UA systems fit into more than one category. For example, both private gardens and community gardens may exist as rooftop gardens, and orchards may exist within community gardens. They may be cultivated by an individual owner or by a community. The various types of UA that exist are important toward providing the planned vegetative diversity necessary to support other associated biodiversity within cities (Figure 1). In this dissertation, I primarily use community or allotment gardens as my study system.

Urban agriculture as important areas of food provision in cities

Urban agriculture is increasingly supported within and around cities due to food security concerns. Several US cities contain *'food deserts'*, where access to fresh

produce is limited due to reduced proximity to markets, financial constraints, or inadequate transportation (Thomas 2010; ver Ploeg *et al.* 2009). For example, assessments of the Oakland, CA food system have underscored that affordability is the most important factor that influences where low-income residents shop for food (Wooten 2008), and residents' limited access to transportation to grocery stores is another fundamental constraint to accessing healthy food (Treuhaft *et al*. 2009).
UA has rapidly increased in developing countries all over the world, especially after the 2008 increase of global food prices (FAO 2014). In many African nations, for example, the percentage of low-income urban population participating in UA has grown from 20% in the 1980s to about 70% in the 2000s (Bryld 2003). This is because UA can be very productive, providing an estimated 15–20% of the global food supply (Hodgson *et al.* 2011; Smit *et al.* 1996). For example, UA provides 60% of the vegetables and 90% of the eggs consumed by residents in Shanghai, 47% of the produce in urban Bulgaria, 60% of vegetables in Cuba, and 90–100% of the leafy vegetables in poor households of Harare, Zimbabwe (Lovell 2010).

Additionally, as urban crop cultivation can also provide significant dietary contributions, communities around the world are using it to improve the health of urban residents. Many successful UA programmes have increased the food security of local residents. For example, New York City's (NYC) *Green Thumb* has become the largest community gardening programme in the US, with more than 600 gardens that support 20,000 urban residents located in ethnically and culturally diverse neighbourhoods where a wide range of community members cultivate and manage the gardens (Lovell 2010). Ongoing expansion in Detroit's urban gardening scene is expected to produce 31% of the vegetables and 17% of the fruits currently consumed by city residents on just 100–350 ha of land (Colasanti and Hamm 2010). Private gardens also contribute significantly to local food production and food security. A study in Chicago showed that the food production area of home gardens was almost threefold that of community gardens. This suggests that home food gardens can contribute significantly to enhancing community food sovereignty (Taylor and Lovell 2012).

Urban agriculture can support high levels of biodiversity

Urban agriculture is an increasingly important urban green space in which to support and enhance urban biodiversity. If designed carefully and deliberately, urban gardens can support high levels of biodiversity and ecosystem services, which in turn allow for more resilient food production systems. Thus, it is important to evaluate that design and management factors that maximize biodiversity and ecosystem services coming from gardens so that communities can be best served by these spaces.

Urban agriculture in the context of urban biodiversity research

Urbanization has been shown to be a force of biotic homogenization where species assemblages across cities become more similar because the similar challenges of the built environment across cities (habitat fragmentation, pollutants, etc.) select for species that can survive and thrive in these systems (e.g. pigeons) (McKinney

2006). This type of selection fundamentally changes patterns of regional biodiversity (species distribution, competition, etc.) (Schwartz *et al.* 2006) as the urban matrix increasingly dominates the landscape. However, research in urban systems over the past decades has studied how local and landscape structure can support ecologically significant biodiversity, like insects, plants, and birds, among others (Beninde *et al.* 2015). For example, studies on urban insect pollinator community richness, pollination and pollinator meta-populations have unveiled how organisms interact within local habitats and respond to the urban matrix (Jha and Kremen 2013; Lowenstein *et al.* 2014).

While research has studied ecological interactions in urban gardens, many studies often focus broadly on all green spaces (e.g., parks, hedgerows, cemeteries) (Andersson *et al*. 2007), thereby obscuring the specific ecological dynamics and importance of urban agriculture in its contributions to urban biodiversity. In this dissertation, I focus exclusively on urban agricultural systems to fill this research need. As part of the green infrastructure of urban landscapes, UA can exhibit a wide breadth of biotic diversity and provide critical resources to species sensitive to detrimental side-effects of urbanization, thus combating the homogenization effect. As a preface to the topics researched in this dissertation, in the following sections, I describe how specific local and landscape habitat characteristics, as well as human characteristics of urban neighbourhoods, influence biodiversity and associated environmental and cultural functions such as food provisioning. This will provide critical information to understand my research framework and the results from my work.

Local and landscape factors affect biodiversity in urban agriculture

Both local factors (i.e., habitat characteristics) and landscape factors (i.e., surrounding landscape features) affect the degree to which agroecosystems contribute to biodiversity and ecosystem services like food provisioning (Altieri 1999; Tscharntke *et al.* 2005), and can be applied to urban environments (Angold *et al.* 2006). Local factors include vegetative diversity, abundance of crops and flowering plants, and soil management practices. Landscape factors can include landscape connectivity, landscape diversity within a reference area, and the position along a rural to urban gradient. Yet, our understanding of how these factors interact at both levels with one another to affect biodiversity and ecosystem function is still relatively limited in the urban context (Angold *et al*. 2006; Matteson and Langellotto 2010). This dissertation investigates local and landscape drivers of biodiversity, aiming to fill this gap and contribute to our understanding of the importance of flora and fauna of urban agroecosystems in the context of landscape differences.

Local factors: management and environmental heterogeneity

Habitat management in urban agriculture by vegetative and soil management supports local biodiversity, ecological interactions among organisms (i.e., food webs), and food production for people. Habitat size and quality is a key driver of urban biodiversity (Beninde *et al.* 2015) like beneficial insects (Angold *et al.* 2006; Pardee and Philpott 2014). Urban vegetation management can strategically aim to increase habitat quality for ecological communities in urban agroecosystems because plants provide a 'template' for ecological community formation and species interactions (Faeth *et al.* 2011). Urban gardens can harbor rich floral and ornamental plant communities, providing nectar and trophic resources to support beneficial insect populations and the overall species diversity (Colding *et al.* 2006), as well as high diversity of flowering vegetable and fruit crops, reflective of the cultural diversity of community gardeners (Baker 2004).

It is thus little surprise that larger urban gardens with greater flower and plant abundance and diversity have been related with increased beneficial insect abundance and species richness. Bee community richness increases with urban garden size (Frankie *et al*. 2005), and floral and plant abundance and richness (Frankie *et al*. 2005; Matteson and Langellotto 2010; Pardee and Philpott 2014). Smith *et al*. (2006) also found that solitary bee diversity in urban agroecosystems was positively correlated with presence of certain vegetative components, such as trees, and overall structural complexity. Additionally, Bennett and Gratton (2012) found that high Hymenopteran parasitoid abundance was best explained by increased flower diversity within urban green spaces, which is a finding that has implications for pest control services in urban gardens.

In response to growing concerns over bee populations and pollination services and public popularity, studies have focused on urban bee pollinator responses to local vegetation composition in gardens with implications for ecosystem function and services. In comparison to other urban green spaces, urban community gardens with high ornamental and flower diversity are critical habitats for urban bees, and most studies have found strong correlations between local vegetation characteristics and pollinators. Colding *et al*. (2006) assessed community and domestic gardens in Stockholm, Sweden, and found that community gardens had a high abundance and diversity of flowering plants (over 400 species) that in turn supported a high abundance and diversity of urban pollinators. Urban gardens have been found to harbor both increased abundance and diversity of bumble bees in response to greater flower presence and richness (Andersson *et al.* 2007, Ahrne *et al.* 2009), and overall bee species diversity in response to crop plant and ornamental diversity (Matteson *et al*. 2008). Additionally, Matteson and Langellotto (2010) found strong relationships between butterfly and bee diversity and local floral resources as well as wild areas within urban gardens in New York City. Similarly, Chicago's neighbourhood bee abundance and richness increased as a response to floral diversity and also to human presence, suggesting pollination services are mediated by residents planting a diversity of flowering plants (Lowenstein *et al.* 2014).

The effect of native plants within urban gardens on beneficial insect populations and ecosystem function is still debatable. Matteson and Langellotto (2011) studied the impact of native plant additions on bee, butterfly and predatory wasp species richness in urban community gardens previously dominated by exotic flowers in New York City. They found that increasing native plants in urban gardens did not attract more insect visitors or contribute to visitor diversity or abundance. However, bee presence and abundance has been linked to native plants that are present in urban and suburban areas (Frankie 2005), and increasing native plantings of flowers has indeed been shown to strongly enhance native bee populations in urban community gardens from 5 to 31 species in 3 years (Pawelek *et al.* 2009). Along those findings, Pardee and Philpott (2014) found that native plants provide floral resources for native bee populations in a resource-poor urban landscape in Toledo, Ohio, and stress the importance of native plants in urban gardens to support urban arthropod diversity and abundance, and ecosystem functions like pollination. Interestingly, they found that bee community composition was significantly different between native and non-native gardens, providing evidence that plant community composition influences pollinator community composition.

These differences in response to native plantings may be due to several factors. First, exotic species can provide greater nectar resources for butterflies and bees, thus attracting and fostering biodiversity. Secondly, a significant increase in pollinator abundance and diversity may require larger and more diverse native plant additions, as well as longer sampling periods. Matteson and Langellotto (2011) were limited to a 16-month study, while Pawalek *et al.* (2009) observed results of planting manipulations over a 3-year duration. Third, urban context and the degree of land-use disturbance intensity and frequency may be a driver of observed differences (Pardee and Philpott 2014). In this comparison, New York City's insect biodiversity may be comprised of more generalist feeders (Matteson and Langellotto 2011) or other life history characteristics adapted to more intensely managed and disturbed land. Future assessments will require a standardized research methodology in both scale and sampling duration to determine the role of native planting manipulations in influencing biodiversity and abundance of beneficial insect populations, and should compare and contrast floral nectar resource availability. These research efforts could be greatly benefit from long term partnerships between researchers and the community of practice, including garden organizational leadership and participants as well as city parks and recreation services. In sum, garden practitioners and local to city-wide management practices can support urban insect populations important for ecosystem function and services in urban agriculture for increased food provision.

In both chapters 2 and 3, I contribute to this investigation and discussion of the importance of local management, particularly of floral resources, for ladybird beetles, parasitoid wasps, and pest control services in urban gardens. I investigate these local factors in the context of different landscape factors surrounding gardens, a topic to which I now turn.

Landscape factors: structure and connectivity in the urban landscape

Urban agriculture is distributed across a complex urban landscape. Land-use configuration can enhance or block ecological functions within local ecosystems as built environments generally result in increased impervious cover and fragmentation of urban green spaces, leading to habitat area decline and a reduction in species diversity (McKinney 2006). Fragmentation has been shown to negatively impact urban insect and arthropod populations such as pollinators (Cane *et al.* 2006).

The declines in urban bees are likely a function of increased impervious cover, as increasing impervious cover decreases habitat area, and bee foraging and dispersal movement (Jha and Kremen 2013). Bumble bee diversity follows this trend and was found to decrease in response to increasing landscape impervious cover (Ahrne *et al.* 2009). Increased mobility and dispersal of functional insects is a result of landscape matrix permeability, which is influenced by degree of urban development (or amount of impervious cover), overall complexity of the landscape, and habitat connectivity. Lin and Fuller (2013) equate urban landscape mosaics with agricultural landscapes, as they can be similar in both their homogenization and intensity of land-use. In rural agricultural landscapes, landscape-level intensification can have negative impacts on beneficial insects, and pest control and pollination (Tscharntke *et al*. 2005). This has been shown for urban natural enemy abundances; parasitoid wasp abundance declines as the percent of urban green decreases and impervious cover increases (Bennett and Gratton 2012). Pollinator population abundance also decline in response to urban development intensity (Jha and Kremen 2013). Thus, natural areas can provide source populations and resources to urban agricultural systems to increase the abundance and diversity of functional organisms like bees (Hernandez *et al*. 2009).

To confront negative impacts of fragmentation and urbanization, connectivity and the creation of ‘green corridors’ have been proposed to enhance abundance, diversity, and ecosystem function within and among urban green spaces (Rudd *et al.* 2002). Consistent with this approach, various urban conservation programs have focused on creating green pathways via planting flowers and native grasses in utility easements, hedgerows, riparian corridors, and backyard gardens, to support urban wildlife and add ecological value to cities (Rudd *et al*. 2002). This supports UA biodiversity while also allowing UA to be a critical node of these green corridors. Few have rigorously assessed the efficacy of enhanced connectivity for urban agriculture insect diversity and agroecosystem function. Rudd *et al.* (2002) show that urban gardens can facilitate functional connectivity of urban green spaces, and suggest gardening as a tool to enhance regional habitat quality. Colding *et al.* (2006) demonstrate the importance of garden connectivity within a fragmented and heavily developed urban landscape, with evidence that allotment garden networks support urban metapopulations of native bees by facilitating movement and enhancing pollination function. Thus, urban agriculture has the potential to significantly contribute to the overall green space connectivity and should be integrated into conservation and planning models to increase urban biodiversity and maintain ecosystem services. Further, local management practices can increase habitat quality within gardens and result in high abundance and species richness of insect populations. High quality habitats with high local biodiversity and ecosystem functions can potentially have a ‘spill over’ effect across a landscape when high degrees of landscape connectivity and permeability exist. Thus, connecting urban gardens to existing forms of green corridors at the landscape level can potentially enhance habitat configuration and permeability for mobile biodiversity, and in turn increase insect species abundance, richness, and dispersal to support urban populations and ecosystem functions like pollination. In chapter 3, I experimentally test this hypothesis by measuring dispersal of ladybird beetles from gardens embedded within high to low quality landscapes of differing natural land cover.

Social systems can affect urban agriculture management and food provision

Urban systems are heavily influenced by the environmental conditions (e.g. built environment, changes in climate, changes in water flows), as well as by social conditions (e.g., planning, finance, community attitudes and desires). These conditions also exist within urban gardens, where plant selection, management, and soil preparation are highly affected by the social complexity of networks, organizations, knowledge flows, and power dynamics of gardens and their communities. The nuances of these aspects within the community can drive the motivation, values, and interactions of individuals to influence the management of these spaces and the associated biodiversity and ecosystem service generation (Andersson *et al.* 2007).

The biodiversity of urban agriculture is often ‘infused’ with the human diversity and multipluralism of metropoles (Baker 2004). In an examination of the community gardening movement in Toronto, Canada, Baker (2004) found that elderly gardeners use specific agricultural techniques developed as farmers in rural China to grow culturally appropriate foods for themselves. In central California, Corlett and colleagues (2003) found that urban garden biodiversity and ethnobotany reflects the origins of urban farmers: nearly all of the 59 species of plants reported by Hmong farmers had a cited use in South-east Asian literature for food, seasoning, or medicine. These examples demonstrate that ethno-cultural diversity is reflected in agricultural practices and agrobiodiversity in urban agriculture.

Uneven patterns in urban development leave behind very heterogeneous landscapes and heterogeneous patterns in socioeconomic gradients (Swan *et al.* 2011). Social and economic variation as a result of income inequality can drive urban plant species diversity to influence associated biotic communities (Hope *et al*. 2008). Termed the '*luxury effect*', as urban neighborhood wealth (median family income) increases, plant species diversity can also increase in urban areas (Hope *et al*. 2008). The luxury effect can have bottom-up influences on higher trophic levels within the ecological community, such as urban park and neighborhood avian diversity (Kinzig *et al*. 2005). This suggests that residents within neighborhoods of lower socioeconomic status are experiencing inequitable access to biodiversity-rich urban environments and suggests an inequitable distribution of ecosystem services. Further, this trend is also documented in urban agriculture: in a recent study Clarke and Jennerette (2015) examined the relationships among indicators of economic wealth, human ethnic diversity, and plant crop diversity in Los Angeles community gardens.

They found that ornamental flower species diversity and abundance in urban community gardens per garden plot significantly increased with neighborhood wealth. The authors also found significant trends among dominant gardeners’ ethnicity and species composition: gardens categorized into predominant ethnicities (Non-immigrant, African-American, Asian, and Hispanic) were self-similar in their species composition of food crops and ornamental species compared to gardens of different ethnic groups. This suggests that pollination services may be different considering the importance of ornamental flower diversity and composition for pollinator guilds. The correlative results call for more information on individual motivations, knowledge and values, and social networks that may influence garden biodiversity management. Urban agriculture leads to not just increased urban food production, but increased social interactions; in allotment gardens, gardeners often exchange ecological and cultural knowledge and experience with other gardeners (Saldivar-Tanaka and Krasny 2004) to influence personal management practices (Andersson *et al*. 2007). Thus, there can be a tangible biodiversity spill over not only from plot to plot, but a spill over in the ecological knowledge and learning from gardener to gardener. I examine and expand on the role of this knowledge exchange as one of the key benefits that gardeners report receiving from gardening in chapter 8.

Beyond food provision: The socio-ecological benefits and challenges of urban agriculture

Urban agricultural is common across continents with urban gardens covering hundreds of hectares in Amsterdam, Montreal, Beijing and Barcelona, amongst many other cities (reviewed in Lovell 2010), and such green spaces serve many environmental and social uses for urban citizens. UA is regarded as an important feature for the long-term support of urban systems at global scale (Barthel and Isendahl 2013), and thus critical to the sustainability and resilience of cities. Additionally, with many benefits to cities, urban policy and development have been increasingly adopted to introduce and maintain such systems (McClintock and Cooper 2010). However, some challenges are associated with agricultural systems in cities with many interests competing for land use.

We have tangentially described many of the benefits related to urban agriculture. Researchers and popular media have highlighted the social-ecological benefits and multi-functionality of urban agriculture (Lovell 2010). Urban agriculture is associated with forms of civic agriculture and food justice, community development and social networks, and urban greening and recreation. These benefits are a reason why urban agriculture has been encouraged within alternative agri-food movements. First, civic agriculture emphasizes the localization of food production, and embeds the agri-food system within the social, economic, and ecological systems of a place (Lyson 2004). As a form of civic agriculture, urban agriculture increases urban food security, fresh food access, public health, and food sovereignty. Household participation in community gardens increases fresh fruit and vegetable intake among participants as observed in Flint, MI where gardeners were 3.5 times more likely to get 5 servings of fruits and vegetables daily (Alaimo *et al.* 2008). Further, urban agriculture may offer an opportunity for farmers to grow culturally appropriate, high quality and diverse foods unavailable at the store (Baker 2004), and to utilize their agricultural knowledge to define their own diets (Minkoff-Zern 2012). Urban agriculture offers a suite of other social benefits that may be more important than the actual food growing. A space for daily socializing, community bonding, education, and special events are all well documented benefits of community gardens (Saldivar-Tanaka and Krasny 2004). In New York City, community gardeners value gardens as spaces for reading, writing, and studying in addition to skill-based workshops to learn about farming/cultivation practices, cooking and nutrition (Saldivar-Tanaka and Krasny 2004). Moreover, urban community gardens are also sites of community and citizenship where women form community based on ethnicity and knowledge sharing (White 2010), and the shared experience of adapting to a new country (Corlett *et al.* 2003). In chapter 8, I add to our understanding of garden social benefits by asking how the context of the social and biophysical landscape may influence the specific well-being benefits derived from gardens.

Yet, urban agriculture projects also face many challenges as a community and social movement. Projects confront logistical barriers such as land access, soil contamination from previous industrialization, and lack of water (Guitart *et al.* 2012). Projects also confront structural barriers like accessing and maintaining property rights (Irazabal and Punja 2009). Thus, urban agriculture projects struggle to be sustainable long-term efforts due to compounding challenges related to land security, gentrification, capital and human resources. Further, many projects may not address social justice or issues of race and inequality present in the alternative agri-food movement (Reynolds and Cohen 2016). In sum, urban agriculture can be considered spaces of ecosystem services and ecological wealth (e.g., food, pollination, biodiversity), yet simultaneously spaces of ecosystem disservices (e.g., invasive species or nutrient run-off) and social injustices (e.g., inequitable distribution of resources and environmental pollutants). The interplay between services and disservices has challenged researchers to understand the ecological and social complexities of these systems, and how social-ecological interactions spill over across the urban landscape. In chapter 9, I investigate some of these challenges around water and land use and access in gardens during a time of resource stress from drought. In revealing some of the particular challenges of gardens and the way in which gardeners as individuals and as a collective respond, I complicate the assumption that gardens are egalitarian spaces of social and ecological sustainability due to the external pressures from increasing urbanization.

Supporting urban agriculture and its contributions to city life

To understand the myriad of benefits that UA provides, it is imperative that we protect and maintain these green spaces in rapidly densifying cities. Recent studies have revealed relationships among biodiversity, ecosystem functions and services and several local, landscape, and social factors showing that:

- Biodiversity in UA systems is highly human managed. As vegetation structural complexity and composition in urban agroecosystems is a result of local management, insect diversity for ecosystem functioning can be human-mediated.
- Landscape heterogeneity and increased permeability via increased green space can affect insect species movement, and local biodiversity and community composition, affecting the biodiversity and ecosystem function of UA systems.

Though many of the studies highlighted here have examined local and landscape factors in urban systems, few have looked at both specifically in the urban agroecosystem context to evaluate their relative importance on both abundance and species richness of functional species. Supporting research in urban agriculture that assesses what local, landscape, and social factors affect specific UA ecosystem functions is necessary to develop policies that promote UA systems (Fig. 2). Integrative and multiple approaches can determine how landscape quality surrounding urban gardens can promote species mobility and population numbers, ecosystem service multiplicity across space, and how to facilitate urban garden networks to increase ecosystem function. Experiments like in chapter 3 that test local and landscape factors for species dispersal and fidelity to an urban agroecosystem are important for increasing our understanding of habitat use and arthropod movement in the urban matrix.

While urban pollinators and their services is well studied, there is a gap in our understanding of other ecosystem functions and services in urban agroecosystems. Urban research that mirrors rural agricultural studies can provide information on how an urban matrix and an agricultural matrix compare in functional insect responses to habitat. Additionally, urban gardens may provide other ecosystem services that are not insect-related, including carbon sequestration and storage and water conservation by vegetation and soil (Davies *et al*. 2011). Thus, urban agroecosystems harbor not only biodiversity essential for ecosystem functions, but can be beneficial for climate regulation and resource conservation. In chapters 6 and 7, I provide empirical findings from two urban regions (California, Australia) that show that local and landscape land cover influence garden climate regulation and that climate changes can influence gardening behaviour. Moreover, I show that there is an effect of temperature on the species richness of cultivated plants in gardens. This suggests that temperature variability may be an environmental filter for crop selection and success. However, temperature variability may just change gardeners' watering behaviour, and not their planting strategies.

The influence of management and incorporation of stakeholders in urban gardens is often not studied in tandem with ecological research, despite the fact that local management by residents and organizations have wide implication for increasing ecosystem resilience and service provisioning (Colding *et al.* 2006). A large portion of urban studies end by addressing the role of citizen management in mediating resource-providing habitat to set the stage for ecosystem functions (Andersson *et al*. 2007; Lowenstein *et al*. 2014). For example, creating an urban bee-friendly habitat should involve the participation of community gardens to develop conservation initiatives that focus on creating a garden vegetation structure shown to be correlated to diverse bee communities (Hernandez *et al*. 2009). Thus, a high priority for urban ecologists should be to communicate with and involve stakeholders early in the research process both to develop context relevant questions, and to provide applicable information to garden practitioners and urban planners concerned with creating supportive agroecosystems. Urban gardens, farms and forms of agroecosystems have the opportunity to be areas of biodiversity conservation and to increase local food security. Connecting ecological research on urban agriculture with practitioners can facilitate realistic conservation efforts of urban biodiversity and ecosystem processes that benefit urban social and ecological communities. For this reason, in chapter 6 I use citizen science research methods to collect water use data with the hope that the knowledge generated from the research project is informed by diverse backgrounds, and can be directly applied by practitioners involved in producing knowledge and solutions to urban agriculture challenges.

Dissertation preface

In preface, this dissertation examines relationships among garden biodiversity, local management diversity, landscape diversity (use, change, history), and human diversity (cultures, experiences, knowledge) to better understand the ecology of urban gardens. In the following 10 chapters, divided into three parts, I use ecological theory from natural, rural agricultural, and urban systems in tandem with social theory from urban political ecology and human geography to advise my research approach and analysis. One strength of the interdisciplinary approach is the ability to look at multiple study sites nested within multiple geographic regions (three counties) to support generalizations and system-wide patterns. Yet I will also pay attention to specific contexts to delve into specific research questions (Chapters 3, 6, 9). In this regard, my approach will intertwine gardens as comparable social and ecological systems, but also extract certain garden contexts to highlight socioecological complexity.

In summary, the goals of this work are twofold. First, I aim to contribute to the agroecology, urban ecology, and political ecology literature with needed studies that explain relationships among ecological and social complexities in agroecosystems. Second, I aim to inform urban management to increase environmentally and socially sustainable outcomes. This is needed because although urban agriculture is growing in popularity to increase fresh and sustainable food access, gardeners still lack practical information that may allow them to better manage their garden plots towards ecological and social benefits (Surls et al. 2014).

In the chapters that follow, I draw from multiple urban and agricultural theories, mix ecological and social methodological approaches, and focus on different study organisms - insects, plants, people, soils - to understand the biodiversity, ecosystem services and sustainability potential of urban gardens.

Table 1-1. Different typologies and descriptions of UA systems to highlight the diversity of urban farming.

Type	Description	References
Community or allotment gardens	Represent small-scale, highly-patchy and qualitatively rich (vegetatively complex and species rich) agro-ecosystems that are usually located in urban or semi-urban areas for food production.	Colding *et al.* 2006
Private gardens	Primarily located in suburban areas and may be the most prevalent form of urban agriculture in cities. For example, privately owned gardens cover an estimated 22–27% of the total urban area in the UK, 36% in New Zealand, and 19.5% in Dayton, Ohio, USA.	Loram *et al.* 2007; Mathieu *et al*. 2007
Easement gardens	Gardens often regulated by the local government but located within private or community properties. Urban easements are established with the purpose of improving water quality and erosion control, but they can include a wide array of biodiversity, including food plants, depending on management type. Gardening on verges may also be done as a form of '*guerrilla gardening*' where local communities garden on small patches of soil when few unpaved spaces are available.	Hunter and Hunter 2008; Hunter and Brown 2012
Rooftop gardens or **green roofs**	Any vegetation established on the roof of a building and can be used to improve insulation, create local habitat, provide decorative amenity, and cultivate food plants.	Whittinghill and Rowe 2012

Urban orchards	Tree-based food production systems that can be owned and run privately or by the community. Increasingly, schools and hospitals are establishing fruit trees that provide crops, erosion control, shade and wildlife habitat, and producing food for the local community.	Drescher *et al.* 2006
Peri-urban agriculture	Usually exists at the outskirts of cities that largely serve the needs of the nearby urban population. Typically, these are multifunctional agricultural systems that include a large variety of activities and diversification approaches and contribute to environmental, social and economic functions.	Zasada 2011

Figure 1-1. Some of the diverse forms of urban agriculture: a) rooftop restaurant garden in San Cristóbal, Chiapas, MX; b) campus farm at the University of California, Santa Cruz, CA; c) City Parks & Recreation garden in San Jose, CA; d) Non-profit garden with “adopted” beds in Salinas, CA.

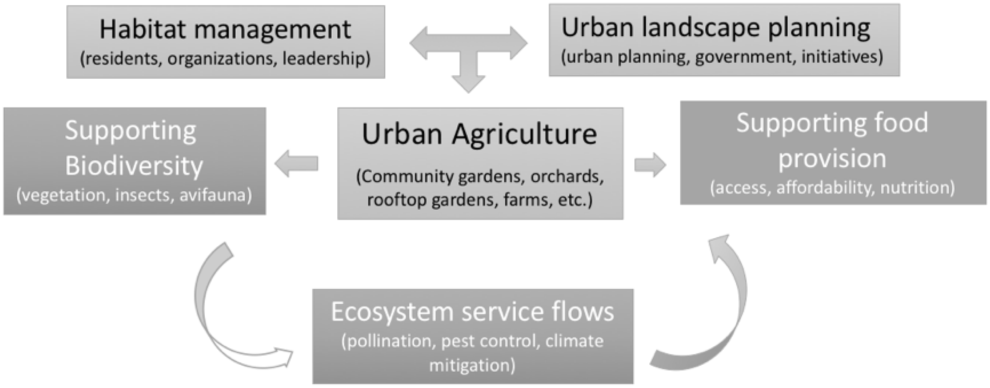

Figure 1-2. Urban agriculture involves a diversity of stakeholders that in turn influence how urban agriculture supports the conservation of urban biodiversity, the flow of ecosystem services, and the cultivation of food benefits.

Part I

2. Landscape and local habitat correlates of ladybeetle abundance and richness in urban agriculture

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Published in *Annals of the Entomological Society of America*, 2016

Abstract

Landscape surroundings and local habitat management affect patterns of insect biodiversity. Knowing which landscape and local factors are more important for insect species diversity informs landscape and local scale land management, yet can be challenging to disentangle. We sought to identify 1) which landscape factors surrounding, and 2) which local habitat factors within urban community gardens influence patterns in lady beetle (Coccinellidae) abundance and species richness. We assessed lady beetle abundance and taxonomic diversity, garden habitat characteristics, and the surrounding landscape composition in 19 gardens over two consecutive years. We found that the amount of natural area surrounding gardens at 3 km was the strongest correlate of abundance and species richness. Specifically, gardens surrounded by less natural area (gardens embedded in more urban landscapes) had higher lady beetle abundance and richness. In gardens embedded in landscapes with more amounts of natural land, local habitat features such as ornamental abundance and crop diversity may become more important for maintaining lady beetle abundance and richness. Our results suggest that within more

urban landscapes, lady beetles may aggregate and accumulate in relatively resource-rich habitats like gardens. Thus, urban landscape quality and local habitat management may all interact to shape lady beetle communities within gardens.

Introduction

Local habitat features and landscape surroundings strongly influence different groups of insects in agricultural and urban environments. Local factors that influence insects include vegetative diversity and structure, abundance of crops and flowers, and grower management practices (Landis et al. 2000, Fiedler et al. 2008). In agroecosystems, certain factors are important for maintaining and conserving insect species biodiversity. For example, local factors, such as crop diversity and spatial structure (Root 1973, Andow 1991) and floral abundance and species richness (Rebek et al. 2005), have bottom-up effects on insect species diversity. Insect species respond differently to plant architecture and spatial diversity due to the spatial allocation of resources and species-specific exploitation of plant structures (Brown and Southwood 1983). From an insect predator's perspective, complex vegetative architecture may either increase (e.g., simple architecture) or decrease (e.g., complex architecture) the efficiency of finding food resources (Southwood et al. 1979, Kareiva 1987, Andow 1991). Thus, there is a relationship between vegetative community composition and structure, and insect community composition even within one trophic guild (Aquilino et al. 2005). Landscape factors that influence insects can include landscape connectivity (Hanski and Beverton 1994), landscape diversity within a sample area (Gustafson 1998), and the position along a rural to urban gradient (Mcdonnell et al. 1997). At larger spatial scales, a landscape of more diverse surrounding landscape elements such as the type of land use (e.g., urban, natural, cultivated) and the amount of land use types (Elliott et al. 2002, Gardiner et al. 2009) can affect insect predator biodiversity within agroecosystems.

The interaction between local habitat factors and surrounding landscape factors shapes insect biodiversity and community composition in agroecosystems embedded within agricultural landscapes. Landscape factors and local habitat factors are not mutually exclusive, and instead there is often an interplay between them to determine local (alpha) and regional (gamma) biodiversity (Tscharntke et al. 2012). Research shows that in simple landscapes (i.e., few land-use types), local agroecosystem factors are more important for explaining insect communities, where in complex landscapes (i.e., many diverse land-use types) local factors are usually less significant and landscape factors have a greater influence on community composition and structure (Tscharntke et al. 2005; 2012). In other systems, habitat quality may be more important for explaining species diversity than landscape factors (Fleishman et al. 2002). In sum, this body of research demonstrates that local habitat factors of systems may be more or less important, and their influence may depend on landscape diversity and composition.

Much research has demonstrated the influence of local and landscape factors on insect abundance and species richness in rural agroecosystems, yet there is still a need to understand the coupling of larger spatial factors and local habitat factors that affect insect communities in urban agroecosystems. Urban landscapes can be complex due to the interaction between heterogeneous local habitat management and greater spatial scale land-uses and processes (Burkman and Gardiner 2014). For example, Matteson and Langellotto (2011) found that increasing ornamental flower abundance to boost local habitat quality had little to no effect on bee communities in simple highly developed urban landscapes. Yet Bennett and Gratton (2012) indeed found that high parasitoid abundance was a function of high flower diversity within urban habitats embedded in simple highly developed urban landscapes. Thus, there is still much to learn about how insects respond to the interplay between local and landscape factors in an urban context, and an increased understanding of these relationships can have important consequences for ecosystem service provisioning in agriculture.

Lady beetles (Coleoptera: Coccinellidae) are charismatic components of insect communities in agroecosystems, and provide natural biological control of herbivorous pests (Cardinale et al. 2003, Obrycki et al. 2009), powdery mildew (Sutherland and Parrella 2009), and scale insects (Evans 2009) to benefit agricultural production. Increased lady beetle species diversity increases biological control services as ecological differences among species within communities can improve herbivore pest control, via niche partitioning (Snyder et al. 2006) and species complementarity (Letourneau et al. 2009). Some species introduced into agricultural systems as a biological control agents, such as the multicolored Asian lady beetle (*Harmonia* *axyridis*), may lead to decreased lady beetle diversity within communities and lower biological control over time (Roy et al. 2016). Thus, because lady beetle biodiversity (abundance and species richness) may act as an insightful proxy for biological control services, we sought to determine how lady beetle communities in urban agroecosystems (i.e., community gardens) are affected by surrounding landscape factors and local habitat factors. We sampled urban gardens that vary in landscape and local features across three spatially distinct regions of California to address three research questions: 1) What urban landscape factors surrounding gardens correlate with greater lady beetle abundance and species richness? 2) What local vegetation and habitat factors within urban gardens correlate with greater lady beetle abundance and richness? 3) Are landscape or local factors stronger correlates of lady beetle abundance and species richness in urban gardens? In understanding these relationships, we seek to provide information for urban agriculture management and landscape-scale urban land management approaches that may increase and conserve urban lady beetle biodiversity across the urban landscape, and potentially promote biological control services.

Materials and Methods

Study Region

This study took place in 19 urban gardens in three counties (Santa Clara, Santa Cruz, and Monterey) in the central coast region of California, USA. The gardens differ in local habitat (structural and compositional diversity of both crop and noncrop species) and landscape context (amount of natural, agricultural, and urban habitat in the surrounding area). All gardens have been cultivated for 5–47 years, range from 444 to 15,525 m^2 in size, and are each separated by at least 2 km (Fig. 1). All of the gardens use organic management practices and prohibit the use of chemical pesticides and insecticides.

Data Collection

We sampled lady beetles in 20- by 20-m plots at the center of each garden six times during 2014 (17–20 June, 7–10 July, 27–30 July, 19–21 August, 8–10 September, 29 September–1 October) and six times during 2015 (16–19 June, 7–10 July, 31 July–1 August, 11–14 August, 1–3 September, 21–24 September). We sampled lady beetle adults with two common methods, visual surveys and sticky traps (Finlayson et al. 2008, Gardiner et al. 2009), in order to assure that a larger fraction of the lady beetle community was sampled. First, we visually surveyed and collected lady beetles in eight randomly selected 0.5- by 0.5-m plots within the 20- by 20-m plots. In each 0.5- by 0.5-m plot, we searched all herbaceous and nonherbaceous vegetation and the ground cover (e.g., leaf litter when present) for adults. Here we assumed that lady beetle food sources would be concentrated in the vegetation to attract beetles. Second, we placed four 3” by 5” yellow sticky strip traps (Item 2872, BioQuip Products Inc, Rancho Dominguez, CA) on galvanized wire stakes placed in the ground next to vegetation at four random locations and left them for 24 h. All lady beetles were identified to species—or to genus when species identification was impossible (e.g., *Scymnus* sp. on sticky traps)—using online resources (e.g., Discover Life 2014) and identification guides (Gordon 1985). Specimens are housed in a collection at the Philpott Laboratory at UC Santa Cruz. We pooled abundance and richness data from all visual and sticky traps per site for each sample date to obtain one abundance count and one species richness count per site. In our study, we define and discuss the lady beetle community of each garden as the adults sampled using both visual and sticky trap methods.

On the same dates lady beetles were surveyed we also assessed local structural characteristics and vegetation in four random 1- by 1-m plots within the 20- by 20-m plots. We determined abundance and richness of all herbaceous plants (including crops, weeds, ornamental plants), height of tallest herbaceous vegetation, and ground cover composition (percent bare soil, rocks, litter, grass, mulch; Table 1). In addition, we measured canopy cover at five points in each 20- by 20-m plot, and counted the number and species of trees and shrubs in the plot, and the number of trees and shrubs in flower. We pooled this local habitat data at each garden for each sampling period for each site.

We examined the surrounding landscape composition with data from the 2011 National Land Cover Database (NLCD; Jin et al. 2013). We created four main land use categories including: 1) natural land (combined deciduous forest [NLCD number 41], evergreen forest [42], mixed forest [43], shrub/scrub [52], and grassland/herbaceous [71]); 2) urban land (combined developed low intensity [22], developed medium intensity [23], and developed high intensity [24]); 3) open land (developed open space [21]); and 4) agriculture land (combined pasture/hay [81], and cultivated crops [82]). These categories allowed us to determine the percent of urban, natural, and agricultural land surrounding garden sites and to determine the dominant landscape association or context. We assessed landscape composition within buffers surrounding gardens at 200 m and 3 km. We chose 200 m as a fine-scale landscape variable because it has been defined as the edge of the surrounding landscape matrix in rural agricultural systems to assess lady beetle spillover dynamics (Rand and Louda 2006). We chose 3 km as a large-scale landscape variable because it is the dispersal range cited for many common lady beetle species in California (Gordon 1985) and is similar to other lady beetle studies (e.g., Gardiner et al. 2009). Within each 200 m and 3 km buffer, we used spatial statistics tools in ArcGIS (v. 10.1) to calculate the percentage of each land cover type by dividing the area of each type within a buffer by the total area in each buffer (Environmental Systems Research Institute [ESRI] 2011). Last, we added the percentage of each type for each respective category to determine the total amount of each land cover for each category, within each buffer.

Data Analysis

We used tree structured regression models using the *party* package in R (v. 0.99.489) (R Core Team 2016, Hothorn et al. 2015) to determine what local factor variables and landscape factor variables best explain greater lady beetle abundance and species richness. The tree regression models are a nonparametric class of regression trees that can analyze nominal, ordinal, and numeric response variables like abundance. The tree regression models utilize recursive partitioning by conditional inference, and are appropriate for analyzing data sets with multiple covariates such as our own. Further, these models allow one to visualize the relationship among explanatory variables and the respective hierarchy of importance. We analyzed the data ($n = 204$ observations over 2014 and 2015) using these trees for lady beetle abundance and species richness as our response variables, and landscape factors (i.e., percent of land cover categories) at 200 m and 3 km spatial scales ($n = 8$) and local habitat factors ($n = 17$) as the explanatory variables (Table 1). Next, we isolated aphidophagous species (i.e., predators of aphid herbivore pests; refer to Table 2). Here our intent was to use the presence and richness of these species as a potential indicator for pest control services, as aphids are common pests in gardens. We also examined patterns for lady beetle data without including *Psyllobora viginitimaculata* (a mycophagous species) because it is ubiquitous in the data, and might have skewed other community patterns. We set the models with a minimum criterion of 0.95 (i.e., *P*-value smaller than 0.05), and a minimum value of 20 sum of weights (i.e., number of instances or cases) to implement a split in the tree. We ran the analyses for total lady beetle abundance, total species richness, aphidophagous lady beetle abundance, and aphidophagous species richness as our dependent variables in the model (Table 1). In the analysis, each sampling period per year represented an individual replicate. We chose to analyze each as an individual replicate because in our field observations the vegetative composition (i.e., plants and crops grown) and vegetative structure (i.e., the orientation of plants, the structure of garden beds) changes within the sites month to month in response to temporal variation and changes in management. This analysis aimed to best account for how beetles may respond to this monthly variation.

Results

We found 1,809 individuals of 16 unique species across 2014 ($n = 877$) and 2015 ($n = 932$). The most common lady beetles in our samples included the mildew-eating *Psyllobora vigintimaculata* (71.3% of all individuals), mite-eating *Stethorus* spp. (6.1%), and the aphidophagous *Hippodamia convergens* (5.1%) and *Harmonia axyridis* (5.0%; Table 2). We captured a greater number of individuals via visual sampling methods than by using the sticky trap method. Of those captured, 62.7% of samples came from visual surveys and 37.3% from sticky traps. However, sticky trapping captured a greater number of species ($n = 16$) than visual ($n = 13$) over the course of the 2-yr sampling period. Overall, few species were only observed in a single site (Table 2).

The percent amount of natural land within 3 km had the greatest influence on lady beetle abundance and richness in each model. Total lady beetle abundance was greater in gardens situated in landscapes with less than four percent natural land within 3 km (Fig. 2a; $P < 0.001$). In gardens surrounded by more than four percent natural land within 3 km, lady beetle abundance was greater in sites with greater ornamental flower species richness (Fig. 2a; $P = 0.03$). In gardens with fewer ornamental flower species, abundance was slightly greater in gardens surrounded by more open land within 3 km (Fig. 2a; $P = 0.03$). Lady beetle species richness was most influenced by the amount of natural land within 3 km, with richness greatest in gardens surrounded by less natural land within 3 km ($P = 0.001$), in smaller gardens ($P = 0.02$), and in gardens with fewer trees and shrubs ($P = 0.007$; Fig. 2b). In gardens surrounded by more natural land, richness was greatest in gardens with less mulch and straw ($P = 0.004$) and fewer crop species ($P = 0.03$).

For aphidophagous lady beetle species, natural area within the landscape was also the most influential predictor of increased abundance (Fig. 2c; $P = 0.005$). In gardens surrounded by less natural land, abundance was greater in gardens with more trees and shrubs ($P = 0.03$). Species richness of aphidophagous lady beetles was greatest in gardens surrounded by less natural land within 3 km (Fig. 2d; $P < 0.001$). In gardens surrounded by more natural land, richness was greatest in gardens with less mulch and straw ($P = 0.006$) and more grass ($P = 0.04$).

Discussion

Our study shows that lady beetle abundance and species richness correlate with both local and landscape factors, but that one landscape feature tends to be at the top of the hierarchy. In our study system, the amount of natural land within 3 km (an ecologically relevant scale for lady beetles) had the greatest influence on local abundance and species richness within urban gardens. In particular, we found that there is a split such that different local factors were important predictors of lady beetle abundance and richness depending on the amount of natural land in the landscape.

Our first question was what landscape factors surrounding gardens correlate with greater lady beetle abundance and species richness. We found that landscape factors significantly influence lady beetle communities within urban gardens, and specifically, the amount of natural and urban land present in the surrounding landscape at greater spatial scales. Among our sites, we found that abundance and species richness was greater in gardens embedded in more urban landscapes with very little natural land-use surroundings. This was surprising, as theory may predict that habitats surrounded by less natural land would have fewer species due to a low availability of resources within the surrounding urban landscape matrix (Vandermeer and Carvajal 2001). This suggests that lady beetles may be accumulating or concentrating in gardens in more urban landscapes (i.e., those surrounded by predominantly urban land use) due to a low availability of resources and habitat elsewhere in the surrounding landscape. In these landscapes, the garden may provide the only available habitat refuge. Some lady beetle species may indeed be attracted to urban landscapes; for example, the invasive *H. axyridis* has been found to show a preference for urban habitats like gardens and parks (Roy et al. 2016) to suggest that lady beetle species likely experience the urban landscape matrix differently.

Our second question was what local vegetation and habitat factors within urban gardens correlate with greater lady beetle abundance and richness. Here, we found that certain garden habitat factors correlate with lady beetle abundance and richness, indicating that garden management can provision for lady beetle habitat. In particular, we found that gardens that incorporate different structural elements can positively and negatively influence lady beetle communities. For example, lady beetle species richness increased in the presence of more trees and shrubs in gardens, indicating that increased habitat complexity and structure of annual, long-established vegetation may provide critical habitat for resident species. Further, we found that lady beetle abundance increased as the number of flowers in gardens increased, providing an example of how local management can easily provision for beetles by planting a suite of ornamental flowers or crops that have a high number of flowers. Interestingly, lady beetle abundance decreased in more diverse cropping systems, and richness decreased in the presence of greater mulch and straw ground cover. This may be explained instead by resource concentration in structurally simple habitats, where herbivores are more likely to concentrate in monoculture stands of their host plant and in turn attract predators to these locations (Root 1973, Andow 1991). Thus although this study did not attempt to infer herbivore prey populations and densities within gardens, this finding suggests that certain local factors that were important for lady beetle communities may also influence herbivore (e.g., aphid) populations and therefore be influencing lady beetles.

Our third question was whether landscape or local factors were stronger correlates of lady beetle abundance and species richness in urban gardens. In our study system, our results suggest that landscape factors play a stronger role in shaping lady beetle communities within urban gardens than local factors. We found that local management was less important for gardens that are embedded within more urban landscapes, which suggests that there may be a threshold of the net effect of local management to provision for lady beetle biodiversity in urban landscapes with little land-use diversity (i.e., simple landscapes). This has been found in rural agricultural systems, where local habitat management strategies (e.g., native plantings like hedgerows or flowers) in farms surrounded by monoculture fields with little land-use diversity have little effect on increasing insect species diversity (Tscharntke et al. 2012). A similar trend may be at work in our urban system, where gardens that are surrounded by predominantly urban land-uses are relatively resource-rich habitat in an otherwise resource-desolate urban landscape, and individuals and species may accumulate over time due to high colonization and low emigration. Here, habitat availability may be more important than habitat quality in these simple landscapes, where specific habitat factors such as greater floral abundance and crop diversity are less significant if a garden is located in an intensively developed landscape with little natural vegetation or land-use diversity.

In contrast, gardens that are surrounded by more natural land uses or a diversity of land uses may experience lower abundance and species richness of lady beetles at certain time points due to high mobility and low residency time. Lady beetles aggregate in natural forested or grassland areas to overwinter and reproduce (Hagen 1962) or to escape competition pressures (Gardiner et al. 2009), and gardens near large natural areas may function as resource sites, not residential habitat. This would explain why specific local habitat variables, such as greater ornamental flower richness, were important for explaining greater lady beetle abundance and species richness in gardens surrounded by more natural area. Here, we may find longer residency times in higher resource quality gardens, in which local vegetation complexity and resource availability, like the availability of pollen food resources, can attract individuals to and maintain populations within gardens (Rebek et al. 2005, Lundgren 2009) and relax dispersal processes (Hanski and Beverton 1994, Fleishman et al. 2002).

Last, while our study did not aim to explicitly test species–area relationships in gardens, it is interesting to note the indication of a "small island effect" within highly developed landscapes. Small island effects can occur when environmental disturbance and stochasticity fail to uphold species–area relationships in island biogeography theory (Lomolino 2001, Lomolino and Weiser 2001). The theory has been used to explain increased insect population numbers in highly disturbed urban habitats such as roundabouts (Helden and Leather 2004) and increases in bird species richness postdisturbance in rural agriculture (Ferraz et al. 2003). We found a negative correlation between garden size and species richness, and that gardens situated in intensively developed urban areas harbor high species richness regardless of being large (e.g., >1 acre) or small (e.g., <0.5 acres). The two community gardens where we found the highest species abundance and greatest richness are small, but are some of the only green spaces in the urban centers of Salinas and Santa Cruz, respectively. Thus, these gardens may function as critical habitats in perhaps an otherwise low resource quality urban landscape. In sum, our findings in this landscape show that there is an interplay between landscape and local factors that influence lady beetle communities in gardens, which can have significant management implications at both a local and a landscape scale.

In conclusion, lady beetle species are important for biological control services, controlling crop pests, mildew, and disease. Provisioning for a diverse lady beetle community requires an understanding of how multiple factors at multiple scales affect patterns of lady beetle community composition. Future research should examine the effects of the urban landscape on lady beetle population movement, and explore how gardens within a landscape may be connected via lady beetle dispersal, and how this may be important for urban conservation and urban landscape planning. To conclude, this study demonstrates that for urban growers, a perspective beyond the garden gate to the surrounding landscape may be the first step when strategizing local habitat management for lady beetle communities that provide a suite of ecosystem services for food cultivation in urban agriculture.

Table 2-1. Explanatory and dependent variables used in the analysis.

Explanatory variables	**Scale**	**Min. value**	**Max value**	**Mean**
Garden size	acres	0.11	3.84	1.09
Garden age	years	6.00	48.00	19.11
% Bare soil cover	1 × 1 m	2.00	99.75	41.38
% Grass cover	1 × 1 m	0	33.00	3.49
% Herbaceous plant cover	1 × 1 m	1.25	95.75	50.39
% Mulch/Straw cover	1 × 1 m	0	96.50	25.22
% Rock cover	1 × 1 m	0	50.00	4.90
% Leaf litter cover	1 × 1 m	0	81.75	10.92
Height of tallest vegetation (cm)	1 × 1 m	4.50	261.25	82.75
No. of flowers	1 × 1 m	0	3000.00	139.80

Explanatory variables	**Scale**	**Min. value**	**Max value**	**Mean**
No. of crop spp.	1 × 1 m	0	15.00	5.28
No. of ornamental flower spp.	1 × 1 m	0	7.00	1.57
No. of weed spp.	1 × 1 m	0	14.00	5.15
No. of grass spp.	1 × 1 m	0	3.00	0.86
% Canopy cover	20 × 20 m	0	55.54	0.62
No. of trees/shrubs	20 × 20 m	0	91.00	16.95
No. of trees/shrubs in flower	20 × 20 m	0	28.00	4.80
% Urban land	200-m radius	6.52	100.00	71.82
% Open land	200-m radius	0	72.46	16.91
% Natural land	200-m radius	0	52.17	10.04

Explanatory variables	**Scale**	**Min. value**	**Max value**	**Mean**
% Agricultural land	200-m radius	0	7.19	0.39
% Urban land	3-km radius	14.61	93.95	53.40
% Open land	3-km radius	5.85	25.05	13.82
% Natural land	3-km radius	0.05	58.47	20.67
% Agricultural land	3-km radius	0	23.82	4.20
Dependent Variables				
Lady beetle abundance	20 × 20 m	0	71	8.94
Lady beetle species richness	20 × 20 m	0	9	2.01
Aphidophagous abundance	20 × 20 m	0	21	1.55
Aphidophagous species richness	20 × 20 m	0	6	0.91

Minimum, maximum, and mean indicate cumulative values over the whole sampling period.

Table 2-2. Lady beetle species collected in urban gardens in the California Central Coast between June–October 2014 and 2015

Tribe or Genus	**Species**	**No. of sites found**	**Feeds on**	**Ecological function in agriculture**
Adalia	*Adalia bipunctata*	1	Aphids and mites	Predator/pest control
Coccinella	*Coccinella californica*	8	Mostly aphids	Predator/pest control
Coccinella	*Coccinella septempunctata*	7	Mostly aphids	Predator/pest control
Cycloneda	*Cycloneda polita*	4	Mostly aphids	Predator/pest control
Cycloneda	*Cycloneda sanguinea*	8	Mostly aphids	Predator/pest control
Harmonia	*Harmonia axyridis*	12	Mostly aphids	Predator/pest control
Hippodamia	*Hippodamia convergens*	16	Mostly aphids	Predator/pest control
Hyperaspis	*Hyperaspis quadrioculata*	5	Aphids and scale insects	Predator/pest control
Nephus	*Nephus binaevatus*	1	Aphids and scale insects	Predator/pest control
Olla	*Olla v-nigrum*	1	Mostly aphids	Predator/pest control
Psyllobora	*Psyllobora vigintimaculata*	17	Fungus	Fungus and mildew control

Tribe or Genus	**Species**	**No. of sites found**	**Feeds on**	**Ecological function in agriculture**
Scymnus	*Scymnus coniferarum*	2	Mites and scale insects	Predator/pest and mite control
Scymnus	*Scymnus cervicalis*	1	Mites and scale insects	Predator/pest and mite control
Scymnus	*Scymnus marginicollis*	8	Mites and scale insects	Predator/pest and mite control
Scymnus	*Scymnus nebulosus*	1	Mites and scale insects	Predator/pest and mite control
Scymnus	*Stethorus punctum*	12	Mites and scale insects	Predator/pest and mite control

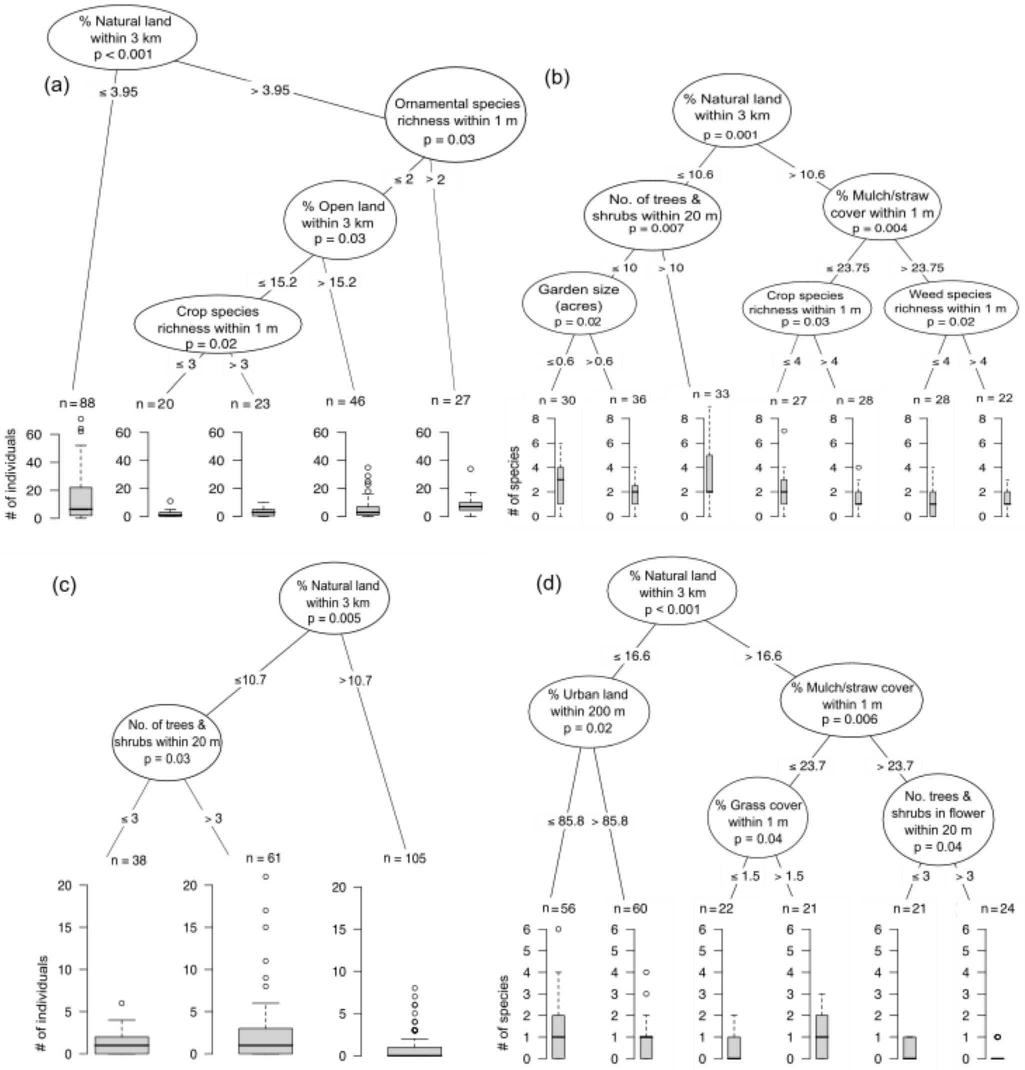

Figure 2-1. Tree structured regression models displaying the landscape and local correlates of (a) abundance of lady beetle individuals, (b) species richness of lady beetles, (c) abundance of aphidophagous individuals, and (d) species richness of aphidophagous individuals in urban gardens.

3. Cityscape quality and resource manipulation affect natural enemy biodiversity in and fidelity to urban agroecosystems

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published in *Landscape Ecology*

Abstract

Context: Complex landscapes with high resource availability can support more diverse natural enemy communities and better natural pest control by providing resources and facilitating organism dispersal. Moreover, in agricultural landscapes, local agroecosystem management can support biodiversity maintenance and pest control by adding resources in less complex landscapes with fewer resources. However, we lack an understanding of how local and landscape factors interact to affect natural enemy communities and their site fidelity to agroecosystems in urban landscapes (i.e., cityscapes).

Objective: To better understand how local and landscape factors influence natural enemies in urban agroecosystems, we used urban community gardens as a model system to test if and how local resource manipulation and differences in cityscape quality affect natural enemy (ladybird beetles, parasitoid wasps) communities and their fidelity to urban habitats.

Methods: We performed two manipulations. First, we added local floral resources in 6 of 12 gardens situated in different cityscapes to measure differences in natural enemy biodiversity. Second, in those 12 gardens, with and without resource additions, we manipulated populations of a common natural enemy, Hippodamia convergens, to assess fidelity to the gardens.

Results: Floral resource additions increased parasitoid abundance and changed community composition, but had little effect on ladybeetle abundance, richness or site fidelity. Rather, ladybeetle fidelity to gardens was lower in gardens in low quality cityscapes with high impervious cover.

Conclusions: Cityscape quality influences natural enemies in and fidelity to gardens. Landscape-moderated biodiversity patterns observed in rural landscapes likely differ from urban contexts with implications for pest control.

Keywords: Landscape composition; dispersal; urban gardens; agroecosystem management; predator; parasitoid

Introduction

Landscape compositional heterogeneity affects population dynamics, biodiversity conservation, and ecosystem services in agroecosystems by influencing the availability and spatial distribution of resources (Denys and Tscharntke 2002;

Rourke et al. 2011). Diverse landscapes with more mixed land cover types are high quality landscapes that generally support a greater diversity of species by providing different resources for different organisms (Tscharntke and Brandl 2004), especially for mobile organisms with complex life histories (e.g., pollinators, natural enemies) (Kremen 2005; Chaplin-Kramer et al. 2011). More simple landscapes with fewer resources, in contrast, are low quality landscapes and may contain high barriers to dispersal and may increase the fidelity of individuals to habitat fragments (Fahrig 2003). Resource availability in the landscape therefore determines landscape quality and drives dispersal and colonization patterns (Schellhorn et al. 2015a), but interactions between landscape quality and local habitat management can influence populations, their dispersal, and service provisioning (Martin et al. 2016). Local habitat manipulation (e.g., through plant resource additions) can increase habitat quality, better support biodiversity and thereby can enhance ecosystem services in simple, low quality landscapes (i.e., the intermediate landscape complexity hypothesis) (Tscharntke et al. 2012). However, the positive effect of local resource manipulation on biodiversity and service provisioning can be relatively less impactful in already high quality landscapes composed of mixed land cover types. This is because high quality landscapes support dispersal between patches and maintain high regional (beta) diversity everywhere; in addition, high quality landscapes can provide spatial insurance in ecosystem function through high beta diversity maintenance if there is a local environmental disturbance or change in local management (i.e., the landscape-moderated insurance hypothesis) (Loreau et al. 2003; Tscharntke et al.

2012; Gámez-Virués et al. 2015). Thus, as for systems in other landscapes, landscape-scale processes can have strong effects on biodiversity, function, and services in agroecosystems.

While local and landscape drivers of agroecosystem biodiversity and dispersal in rural landscapes are increasingly understood (Rourke et al. 2011; Martin et al. 2016), we still need to better understand how local and landscape factors interact in urban landscapes – what we term 'cityscapes'. Cityscapes are increasing in global cover, and are novel in their landscape-scale heterogeneity (in land cover) and in local-scale habitat management (Cadenasso et al. 2007; Kowarik 2011). Local and landscape factors inconsistently affect biodiversity in cityscapes, likely due to landscape-scale habitat loss, fragmentation, and frequent disturbance that in turn affect organism dispersal and use of urban habitats (Angold et al. 2006). For example, in urban gardens, increasing garden vegetation complexity through floral resource addition may (Pawelek et al. 2009) or may not (Matteson and Langellotto 2009) enhance beneficial insect diversity in cityscapes with high amounts of impervious cover (i.e., concrete and built impermeable surfaces) that may be of low quality. The effects of cityscape quality and local management on biodiversity are likely explained in large part by their effects on organism dispersal. For example, high impervious cover may hinder dispersal and may increase site fidelity of individuals to a habitat if emigration is associated with increased mortality risk in a low quality hostile matrix (i.e., low quality land cover in which habitat patches are embedded) (Fahrig 2001). In contrast, cityscapes of higher matrix quality with more natural vegetation cover may facilitate dispersal, resulting in lower site fidelity of individuals to habitats but overall high landscape connectivity through their movement. However, there is little to no information on arthropod population movement in cityscapes or regarding how cityscape quality may trigger or hinder dispersal from urban ecosystems.

In this study, we use urban agroecosystems (community gardens) to test if and how local habitat manipulation and differences in cityscape quality affect natural enemy communities and their fidelity to urban habitats. In our study system, there are strong but variable landscape-scale effects on natural enemy biodiversity (Egerer et al. 2017). Indeed, gardens in low quality cityscapes (i.e., with greater amounts of impervious land cover) generally have higher abundance and species richness of ladybeetles (Egerer et al. 2016), counterintuitive to aspects of island biogeography theory that would predict lower abundance and richness in smaller fragments farther away from other greenspaces (MacArthur and Wilson 1976). Yet for gardens in high quality cityscapes (i.e., with greater natural land cover), local factors like greater floral abundance and greater grass groundcover increase natural enemy abundance and species richness, respectively, likely by providing necessary food and shelter across life stages (Egerer et al. 2016). An interplay among local and landscape factors is affecting natural enemy dispersal behavior and the fidelity to gardens, but we still do not know the mechanisms driving these patterns. This information is significant because factors that affect natural enemy dispersal and site fidelity can affect pest control services (With et al. 2002), and therefore have important implications for improving sustainable pest control through habitat management in urban agroecosystems.

We conducted two manipulation experiments to test if and how differences in cityscape quality and local resource availability influence the abundance, diversity, composition, and site fidelity of natural enemies (ladybeetles, parasitoid wasps) in and to gardens. First, we manipulated garden floral resource availability to ask whether local resource (floral) additions affect the abundance, diversity, and composition of ladybeetle and parasitoid communities. Here, we hypothesized that the addition of floral resources will have a stronger influence on the abundance, richness, and composition of natural enemy communities in low quality cityscapes because added floral resources provide important food (nectar, pollen) and habitat within the garden that is less abundant in the surrounding impervious matrix. Second, we did a mark-recapture experiment of a common ladybeetle species to ask whether local resource manipulation and cityscape quality affect the site fidelity of ladybeetles to gardens. Here, we hypothesized that (i) local resource manipulation will have a stronger influence on ladybeetle fidelity to gardens in low versus high quality cityscapes through local resource provision that slows dispersal; and (ii) high quality cityscapes will facilitate dispersal due to abundant resources in the landscape, and gardens in high versus low quality cityscapes will have lower site fidelity of beetles. Together, the two experiments organized around predictions of landscape moderated biodiversity (sensu Tscharntke et al. 2012) aimed to determine if the effect of local manipulation on natural enemy communities or fidelity varies with cityscape quality.

Methods

Study system

We worked in 12 community garden sites between 197 and 3,656 m^2 in size, separated by at least 2 km, in Santa Cruz and Monterey Counties in the California central coast in May 2017 (Figure 1). The gardens in these regions have similar microclimates (i.e., daily max temp and daily average temp) (Lin et al. 2018) but vary in local management of vegetation and groundcover by gardeners, and in their landscape surroundings. All sites are managed towards the cultivation of organic produce, therefore prohibiting the use of pesticides. The sites are surrounded by different amounts of impervious land cover, agriculture land cover (e.g., crop, pasture), and natural land cover (e.g., forest, grass, shrub). We selected the sites because they exist on either a low or high cityscape quality spectrum. We examined cityscape quality within 2 km of gardens because ladybeetles and parasitoids respond positively or negatively to landscape factors (e.g., amount of impervious cover) within this scale in our system suggesting that this spatial scale is important for their movement (Egerer et al. 2017). We examined the surrounding landscape composition with data from the US Geological Survey's 2011 National Land Cover Database (NLCD) (Jin et al. 2013), and calculated the percent impervious land cover (NLCD classes 23, 24; 30 m resolution) within 2 km buffers surrounding the gardens with spatial statistics tools in ArcGIS (v. 10.1) (ESRI 2011). We classified gardens surrounded by > 80% impervious land cover to be low quality cityscapes, and gardens surrounded by < 30% impervious cover to be in high quality cityscapes (Online Resource 1). This resulted in 6 gardens of each landscape type. The difficulty in replicating the experimental treatments across the region (site availability, time) limited our ability to increase treatment replication numbers, and introduces a limitation to our study.

Phase one: Floral resource addition experiment

In the first manipulation experiment, we tested whether adding local floral resources affects the abundance, diversity, and composition of ladybeetle and parasitoid communities in gardens of low versus high cityscape quality. We randomly assigned 6 of the 12 gardens (3 in each County, 3 of each cityscape quality) to receive a floral resource addition treatment. This resulted in four treatment groups: 1) gardens in low quality cityscapes with floral resource additions; 2) gardens in high quality cityscapes with floral resource additions; 3) gardens in low quality cityscapes without floral resource additions; and, 4) gardens in high quality cityscapes without floral resource additions. For the floral resource addition, we used three insectary plant species: sweet alyssum (*Lobularia maritima*), common chamomile (*Matricaria recutita*), and cilantro/coriander (*Coriandrum sativum*). These flowering plants are commonly grown in urban and rural agricultural systems to attract and support natural enemies of crop pests, including ladybeetles (Family: *Coccinellidae*), parasitoid wasps (*Apocrita*), and syrphid flies (*Syrphidae*). These arthropods use floral resources at varying life stages for food (nectar, pollen) in addition to the prey that they consume or parasitize. All plants were grown under standard conditions in 1 L pots in the Thimann Greenhouse at UC Santa Cruz until flowering.

We assessed the natural enemy community and the floral resource density present in each garden 3 days prior to the floral resource addition (Online Resource 2). To assess natural enemies, we divided each garden into 10 x 10 m grid sections and placed one yellow 3” x 5” sticky card trap (Olson Products Inc.) at the center of each section for 48 h. This meant that larger gardens had more traps than smaller gardens in order to account for garden size. We identified all adult ladybeetles to species on the traps. We identified all adult parasitoid wasps to superfamily on the traps, which does introduce a limitation to our biodiversity assessment. In addition, we visually searched for ladybeetle adults on vegetation and groundcover within a 2 x 2 m area in 8 randomly selected 10 x 10 m sections in each garden. In smaller gardens with less than 8 sections (i.e., > 800 m^2), we randomly selected sections to revisit to visually search in another location within the section. To assess floral density, we established a 20 x 20 m survey plot at the center of the garden and counted the number of total flowers in 8 randomly placed 1 x 1 m quadrats.

On the day of the floral resource addition, we placed species mixtures of 8 flowering potted plants in each of 5 randomly located 1 x 1 m areas for a total of 40 plants within 50 m^2 in each of the six manipulated gardens (Online Resource 1). We added alyssum, cilantro and chamomile in a 2:1:1 species ratio to each garden. Based on floral surveys conducted in each site, we estimate that the floral additions increased floral availability by 1 to 69% in each site (approximately 1,625 flowers were added). We then placed sticky card traps at the center of each 10 x 10 m section in each garden. We returned 48 h later to collect the sticky traps, water potted plants, replace wilting plants with fresh pots, and visually survey for ladybeetles at 8 random locations within the garden (8 of the 10 x 10 grid sections, as above).

Phase two: Ladybeetle population manipulation

In the second manipulation experiment, we tested whether difference in cityscape quality and local floral resource addition influences natural enemy site fidelity to gardens using a mark-recapture experiment of a common native ladybeetle species. We released marked individuals of *Hippodamia convergens* in the 12 gardens 4 days after the floral enrichment. *H. convergens* rely on forest cover and vegetation for habitat, disperse ~3 km, consume herbivorous pests like aphids, and are thus popular natural pest control agents used in agriculture. We purchased live adult *H. convergens* from Northwest Beneficials (Bend, OR) prior to the experiment and stored them at 2 °C for 5 days following company instructions. For the release, we marked ladybeetles with yellow fluorescent insect marking powder (BioQuip item #1162Y) to identify upon recapture (Online Resource 1). The use of fluorescent powders is a common method in mark-recapture studies to assess dispersal of ladybeetles (Baker et al. 2003), parasitoids (Corbett and Rosenheim 1996), and other insects (Kareiva 1985) because it does not significantly affect survival or flight (Naranjo 1990) and thus recapture. We released 35,000 marked individuals (1/2 US gallon, recommended amount for small farms and large gardens by these companies (see e.g., www.arbico-organics.com) in each of the 12 gardens during cool (< 16˚C) and overcast weather conditions so that ladybeetles could acclimate to and experience the garden.

Prior to releasing the ladybeetles, we measured the density of cabbage aphids (*Brevicoryne brassicae*) in the gardens because a lack of aphid prey resources may also lower the fidelity to gardens. Cabbage aphids feed in dense colonies on *Brassica oleracea* plants (e.g., cabbages, Brussels sprouts, kale), shortening crop life in urban agriculture (Flint 2013), and are the most common aphid species in our system. We visually surveyed live cabbage aphids on 8 randomly chosen *Brassica* plants within the 20 x 20 m plot (described above).

We returned to each site after 2, 4, 6 and 12 days following the release to visually survey for marked individuals to assess site fidelity (i.e., the number of individuals that stayed in the garden) as a proxy for dispersal. For the visual surveys, we increased our sampling effort in order to increase recapture probability. We walked along transects corresponding to the garden grids, and stopped every 2-5 m to thoroughly search leaves and groundcover for ladybeetles. We counted all live, marked individuals observed and collected individuals to confirm that they were marked using handheld black lights. Here we assumed that (1) detectability and recapture probability of ladybeetles by researchers was consistent across gardens (we have no reason to believe otherwise), and (2) probability of mortality of ladybeetles over time was consistent across gardens. At the end of the survey we released all counted marked individuals back into the garden. To supplement visual surveys, we placed sticky traps within each of the 10 x 10 m grid sections for 48 h (same methods as floral resource addition). We collected all sticky traps and plants 8 days after the release, and did a final visual survey 12 days after the release. This resulted in a total of three sticky trap surveys and four visual surveys. We added the number of ladybeetles on the sticky traps to the prior visual survey for one recapture count for each time point.

Statistical analysis

Effect of cityscape quality and floral resource addition on abundance, richness and community composition

To determine whether cityscape quality and local manipulation affects the abundance and diversity of ladybeetles and parasitoids, we performed two statistical analyses. First, we used generalized linear regression models (GLMs) to compare ladybeetle and parasitoid abundance and richness in gardens in high versus low quality cityscapes before the experiment. This allowed us to test whether cityscapes categorized as high quality support more or fewer natural enemies. We then used GLMs and a model selection approach to measure the relative effect of cityscape quality and floral resource addition on ladybeetle and parasitoid abundance and richness among gardens of the four groups after the manipulation. This analysis was an effort to identify the model structure that best predicted post-experimental ladybeetle and parasitoid abundance and richness considering (1) initial abundance or richness before the manipulation, (2) non-manipulated floral resource density, (3) floral resource manipulation, and (4) cityscape quality. The mean number of individuals and richness of ladybeetle species or parasitoid superfamily observed per trap per site was the response variable. The pre-experimental mean abundance or richness per trap, non-manipulated floral density, landscape type, and floral resource manipulation and their interactions were the predictor variables. We built global models for each response variable using the glmulti package (Calcagno and De Mazancourt 2010) and used Akaike's information criterion for small sample sizes (AICc) to determine optimal model structure (Burnham and Anderson 2002). If model AICc values were not different from one another (< 2 points), we averaged the top models to obtain conditional average model coefficients. Analyses were completed in the R statistical environment version 3.2.4 (R Development Core Team 2013).

To determine whether cityscape quality and local manipulation affects the community of ladybeetles and parasitoids, we utilized constrained multivariate analysis – redundancy analysis (RDA) – to measure how much the variation in the composition of natural enemy communities is explained by cityscape quality and floral resource addition. We used a constrained method because of our a priori hypotheses about the factors that affect composition (i.e., cityscape quality and floral addition). We created a matrix of the variation in species and superfamily composition, and applied a Hellinger transformation using the vegan package

(Oksanen 2015) in R to standardize abundance across taxa. We used analysis of variance to evaluate the statistical significance of the constraint. To determine whether there were significant differences in ladybeetle and parasitoid community composition in groups before and after the floral resource addition, we used Procrustes analysis using the “protest” function in vegan in R to assess similarity among ladybeetle and parasitoid ordinations, respectively. To determine whether there were significant differences in ladybeetle and parasitoid community composition between gardens within groups, we performed an analysis of similarity test (ANOSIM) using the “anosim” function in vegan in R.

Effect of cityscape quality and floral resource addition on ladybeetle fidelity to gardens

To determine whether differences in cityscape quality and local manipulation affect the site fidelity of released ladybeetles to gardens over time, we used linear mixed-effects models (LMMs) with repeated measures to model the log transformed number of marked ladybeetles recaptured with site nested within survey time point as nested random effects. We built four models and used AICc for small sample sizes to evaluate model fit: 1) ladybeetle fidelity (recaptures) predicted by cityscape quality; 2) ladybeetle fidelity predicted by cityscape quality and floral resource addition; 3) ladybeetle fidelity predicted by cityscape quality, floral resource addition, and non-manipulated floral resource density; and 4) ladybeetle fidelity predicted by cityscape quality, floral resource addition, non-manipulated floral resource density, and the interaction between floral addition and floral resource density. We did not include garden size as a cofactor in the models because it provided a weaker model fit in the preliminary analysis. Moreover, due to a significant correlation between aphid density and cityscape quality, we included cityscape but not aphid density in the models. We performed a separate LMM with repeated measures to model ladybeetle fidelity predicted by aphid density. LMM analyses were performed using the *lme4* package in R (Bates et al. 2015).

Results

Effect of cityscape quality and floral resource addition on natural enemy abundance, richness, composition

Floral resource addition had strong impacts on parasitoids – both in abundance and for community composition – but not on ladybeetles. Parasitoid abundance was greater in manipulated gardens than in non-manipulated gardens ($P = 0.009$; Figure 2c; Table 1) although abundance generally decreased from initial abundance across treatments. We found that the abundance and species richness of ladybeetles and parasitoids were overall relatively greater in gardens in low quality cityscapes than in high quality cityscapes before and after the floral resource addition (Figure 2), but before experimental differences were not significant (Online Resource 3). Ladybeetle abundance was lower in gardens in high quality cityscapes than in low quality cityscapes (Figure 2a), but this was not significant (Table 1). Both ladybeetle richness ($P = 0.02$) and parasitoid richness ($P < 0.02$) were best predicted by greater initial richness before the manipulation and not by floral resource addition or cityscape quality (Table 1).

The floral resource addition significantly changed the composition of parasitoid communities in gardens ($m^2 = 0.47$, $P = 0.006$); the gardens with added floral resources were generally more similar to each other in composition, with relatively higher abundance and richness of superfamilies (e.g., of chalcid and ceraphronid wasps) (Figure 3d). Cityscape quality explained 7.9% of the variation in the parasitoid ordination before the manipulation, while cityscape quality (9.0%) and the floral resource addition (14.1%) and their interaction (9.7%) together explained a total of 32.7% of the variation in the parasitoid ordination after the manipulation (Table 2). The composition of parasitoid communities did not significantly differ between cityscape types before the manipulation ($F = 1.01$, $R^2 = 0.09$, $P = 0.47$; Figure 3c), nor among groups after the manipulation ($F = 1.28$, $R^2 = 0.32$, $P = 0.26$; Figure 3d). Parasitoid communities within treatment groups were not significantly different in composition among one another before the manipulation (ANOSIM: $R = -0.02$, $P = 0.56$), nor within groups after the manipulation ($R = 0.03$, $P = 0.38$). For ladybeetles, cityscape quality explained 9.3% of the variation in the ladybeetle ordination before the manipulation; cityscape (9.2%), the floral resource addition (7.8%) and their interaction (5.1%) together explained a total of 22.1% of the variation in the post-experimental ladybeetle ordination after the manipulation (Table 2). The composition of ladybeetle communities did not significantly differ between cityscape types before the manipulation ($F = 0.43$, $R^2 = 0.04$, $P = 0.81$; Figure 3a), nor among groups after the manipulation ($F = 0.77$, $R^2 = 0.22$, $P = 0.68$; Figure 3b). Gardens within groups were not significantly different in ladybeetle community composition before the manipulation ($R = -0.02$, $P = 0.54$), nor within groups after the manipulation ($R = -0.07$, $P = 0.64$). The Procrustes analysis revealed that the manipulation did, however, weakly significantly change ladybeetle community composition ($m^2 = 0.47$, $P = 0.05$). Thus the manipulation had the strongest impact on parasitoid abundance and an effect on ladybeetle community composition, though the total explained variance (22.1% and 32.7%) indicate unexplained variance not accounted for by the explanatory variables.

Effect of cityscape quality and local resources on ladybeetle site fidelity

Landscape type, but not floral resource addition had strong, significant effects on ladybeetle site fidelity to gardens. Gardens in high quality cityscapes had significantly higher recaptures of marked ladybeetles over time than gardens in low quality cityscapes ($P < 0.001$; Figure 4), and the optimal model structure predicting site fidelity only included cityscape type (Table 3). Counter to our expectations, the floral resource addition did not significantly affect this pattern, nor did the floral density already present (Table 3). Aphid density did not significantly directly affect marked ladybeetle recaptures ($P = 0.75$; Table 3); however, aphid density was significantly higher in low quality cityscapes (Welch Two Sample t-test; $t = -5.4$; $P < 0.001$; Online Resource 4).

Discussion

Cityscape quality influences the fidelity of natural enemies (ladybeetles) to urban gardens, and more so than local resource availability. However, increasing floral resources through floral resource addition increases the abundance of and changes the composition of other natural enemy (parasitoid) communities in urban gardens, which may increase pest control services. We found lower site fidelity of marked ladybeetles to gardens in low quality cityscapes (those surrounded by more impervious land cover). We also found that gardens in low quality cityscapes maintained relatively higher ladybeetle and parasitoid abundance and richness compared to high quality cityscapes (those with less impervious cover and more mixed land use) throughout the experiment. Our results show that natural enemies disperse more quickly from habitat patches in low quality cityscapes, and that these habitat patches have abundant and diverse natural enemy communities. The results suggest that natural enemies may move relatively fast in more impervious surroundings, colonizing relatively high quality habitat patches quickly, but have low fidelity to these habitats. Habitats in low quality cityscapes may therefore have relatively high turnover of individuals and maintain high diversity.

The built environment is less conducive to site faithfulness than we hypothesized. Under the framework of agricultural landscape theory, we hypothesized that agroecosystems in cityscapes with more natural cover were of higher quality for natural enemies and that they would better support biodiversity, facilitate dispersal due to landscape connectedness, and therefore would have lower site fidelity to a garden. Indeed, presence and quality of natural habitat in the landscape both facilitates dispersal and supports higher species richness of natural enemies in agricultural landscapes (Gardiner et al. 2009). Our results show that cityscape effects on natural enemy communities and dispersal differ from the rural context. Drawing from diffusion theory, which predicts lower population densities in land cover that facilitates movement (Schultz et al. 2017), it seems that in our system impervious cover and associated features of the urban environment favor site infidelity of individuals to the garden system. Many (possibly most) organisms move faster in the landscape matrix than in habitat patches (Kareiva and Odell 1987; Schultz 1998; Brown et al. 2017; Lutscher and Musgrave 2017), attributed in part to edge effects. In low quality cityscapes with greater impervious habitat, individuals are more likely to come upon an edge, thereby triggering long range movement to the next high quality patch. Yet in gardens that are in higher quality cityscapes, there may be less of a difference in habitat quality between the garden and the surrounding cityscape, meaning that organisms are more likely to experience an edge less frequently and are thus less likely to undertake large movements. In other words, ladybeetles that leave a habitat patch (e.g., a garden) in a low quality cityscape might move away from the area more frequently or might not find the patch again due to higher flight response. Furthermore, the associated features of urban environments such as thermal, light and noise pollution can also affect insect populations and behavior (McIntyre 2000). Indeed, prolonged warmer temperatures and increased artificial illumination may disorient individuals, extend foraging time, and increase dispersal likelihood (Longcore and Rich 2004) and thereby site infidelity.

Local agroecosystem habitat management variably affects natural enemies. The floral resource manipulation supported greater parasitoid abundance and changed community composition, though abundance and richness were lower after the manipulation. The later result may be due to the removal of individuals from the population with sticky traps, and because individuals were less dispersed in the garden (which our sampling method favored) and more concentrated at the introduced plants. The manipulation had no effect on ladybeetle communities or fidelity, which is surprising because we have found floral abundance to be an important predictor of ladybeetle abundance across sites, particularly in gardens surrounded by more natural land cover (Egerer et al. 2016), and because we observed marked ladybeetles utilizing the plants in the gardens during the study. Given that the floral resource additions only increased floral abundance by less than 10% in some gardens, this may not be enough to trigger differences in ladybeetle site fidelity or movement to or from a garden. However, even small additions of flowers can support greater parasitoid populations, which are sensitive to floral presence in urban habitats (Bennett and Gratton 2012) likely due to the importance of floral nectar for their life history (Ellis et al. 2005; Balzan and Wäckers 2013). Urban gardeners can thus provision for natural enemies like parasitoids and therefore natural pest control with the simple addition of flowering crops that are utilized by both people and insects.

Gardens in low quality cityscapes had significantly more aphid herbivore food resources, but aphid density did not significantly directly affect ladybeetle fidelity to gardens. In urban gardens, plant nutrient and water availability are usually heavily supplemented, thereby reducing resource limitation for herbivores (Raupp et al. 2010), and potentially reducing the effectiveness of natural enemies to control them. Urbanization processes may change the strength and importance of direct and indirect effects on trophic interactions between natural enemy and herbivore (Shrewsbury and Raupp 2000), thus it is still possible that aphid density contributes to the effect of cityscape quality on ladybeetle dispersal through indirect effects. Furthermore, the methodological nature of the study assumed that ladybeetle mortality was similar in gardens between landscape types. Yet cityscape quality might also affect ladybeetle predator abundance, such as spiders, thereby affecting ladybeetle mortality and site fidelity. We observed few instances of marked ladybeetle predation by wolf spiders (Lycosidae), which are more active and diverse in gardens with greater floral abundance and in gardens surrounded by agriculture, but not impervious land cover (Otoshi et al. 2015). Moreover, most ladybeetle mortality from predation occurs at the egg or larval rather than adult stage (Weber and Lundgren 2009). Thus while differences in predation in gardens of different cityscape quality could affect ladybeetle site fidelity, we do not have strong evidence of ladybeetle predation effects driving our results.

Urban gardeners are in need of more information on how to sustainably manage pests, as community gardens often require organic practices that prohibit the use of pesticides (Oberholtzer et al. 2014). Although we did not directly test the effects of site fidelity on pest removal, our results may suggest that gardeners at high fidelity sites may benefit from greater pest control services if ladybeetles consume more pests during their longer residency. On the other hand, gardeners at low fidelity sites in low quality landscapes may have lower pest control by ladybeetles in addition to having higher pest abundance. We cannot definitively link fidelity to greater pest control services, but we can suggest that gardeners in these sites may augment their plots through addition of flowering plants to support parasitoids to potentially increase pest control. Future work that assesses how food web relationships among herbivore pests and natural enemies change in strength and direction with differences in landscape type and local resource availability may impart further insight into management application.

We continue to unravel the mechanisms driving natural enemy community ecology and population dynamics in this system. Our previous work suggested that gardens in high quality cityscapes with greater local resource availability may relax dispersal processes and increase the site fidelity of natural enemies to gardens, while gardens in low quality cityscapes may accumulate species due to high colonization and low dispersal (Egerer et al. 2016). While our first conclusion seems to stand with this presented work, our second conclusion requires reconsideration. Gardens in low quality cityscapes may have high biodiversity, colonization, and site infidelity of natural enemies to suggest that urban agroecosystems have more dynamic, rather than static, populations than previously thought.

Conclusion

The landscape matrix is increasingly recognized as a vital resource for biodiversity (Ricketts 2001) and for supporting ecosystem services provided by mobile organisms (Perfecto and Vandermeer 2002; Schellhorn et al. 2015b). Landscape matrix quality can promote or hinder population movement, habitat colonization, and local and regional extinction probability (Vandermeer and Carvajal 2001). Theory predicts that higher quality landscapes generally beget higher biodiversity maintenance by providing resources over space and time to mobile agents (Kremen et al. 2007). Yet, in low quality cityscapes of high impervious land cover, associated abiotic disturbances, and patchy resource availability, population movement and the site fidelity of organisms to urban habitat patches can change. Using natural enemies in community gardens as a model system, we show how site fidelity in the cityscape matrix may follow a different paradigm. Although gardens within lower quality cityscapes had lower ladybeetle fidelity to them, the maintenance of natural enemy diversity within these agroecosystems surrounded by high impervious land cover may further suggest that urban habitat patches are connected through species dispersal. Therefore, it is important to preserve greenspaces like urban gardens for biodiversity conservation in cityscapes.

Table 3-1. Generalized linear models that best predicted parasitoid abundance, parasitoid richness (to superfamily), ladybeetle abundance, and ladybeetle richness (to species) after the floral resource addition where t_0 is the initial abundance or richness before the floral addition.

Model	**Factor**	**Coef.**	**SE_{adj}**	**z**	**P**	**AICc**	△AIC
Parasitoid abundance	Intercept	0.59	0.82	0.73	0.47	40.73	1.75
	Floral addition (flowers added)	1.52	0.58	2.63	0.009		
	abundance t_0	0.38	0.22	2.20	0.03		
Parasitoid richness	Intercept	0.02	0.11	0.15	0.89	4.28	0
	Richness t_0	0.71	0.13	5.20	< 0.001		
Ladybeetle abundance	Intercept	0.28	0.13	1.93	0.05	15.74	1.57
	Cityscape (high quality)	-0.28	0.23	1.22	0.22		
Ladybeetle richness	Richness t_0	0.60	0.21	2.86	0.02	-16.17	0

Table 3-2. Results of redundancy analyses (RDA) and subsequent variance partitioning for ladybeetle and parasitoid communities, before and after the floral resource manipulation (i.e., floral resource addition to six gardens). Rows show the variance explained by pure and joint fractions of cityscape quality (Cityscape), floral resource addition (Floral addition), unexplained variance (Residuals) and total variance explained by all fractions (Total).

	Fractions	**Variance explained**	
		Ladybeetles (%)	Parasitoids (%)
Pre-manipulation	Cityscape	9.3	7.9
	Residuals	90.7	92.1
	Total	*9.3*	*7.9*
Post-manipulation	Cityscape	9.2	9
	Floral addition	7.8	14.1
	Cityscape + Floral addition	17.02	23
	Cityscape: Floral addition	5.08	9.7
	Residuals	77.86	67.3
	Total	*22.1*	*32.7*

Table 3-3. Linear mixed models (A-E) predicting ladybeetle site fidelity to gardens by cityscape quality (Cityscape), floral resource addition (Floral addition), and non-manipulated floral resources present (Floral density) in gardens. Fidelity was measured as the number of marked ladybeetles recaptured (log transformed) after the experimental release. Day of sampling nested within garden site are random effects. Interactions between terms are represented by x.

Model	**AICc**	**Factor**	**Coef.**	**SE**	**t**	***P***
A. Site fidelity ~ Cityscape	136	Intercept	3.65	0.80	4.55	< 0.001
		Cityscape (high quality)	0.67	0.22	3.00	0.004
B. Site fidelity ~ Cityscape + Floral addition	139	Intercept	3.72	0.82	4.54	< 0.001
		Cityscape (high quality)	0.67	0.22	2.98	0.005
		Floral addition	-0.13	0.22	-0.59	0.56
C. Site fidelity ~ Cityscape + Floral addition + Floral density	140	Intercept	3.84	0.86	4.47	< 0.001
		Cityscape (high quality)	0.71	0.24	2.98	0.005
		Floral addition	-0.09	0.24	-0.35	0.73
		Floral density	-0.05	0.10	-0.54	0.59
D. Site fidelity ~ Cityscape + Floral addition * Floral density	141	Intercept	3.18	0.93	3.41	0.002
		Cityscape (high quality)	1.02	0.28	3.63	< 0.001
		Floral addition	1.30	0.75	1.73	0.09
		Floral density	0.13	0.14	0.98	0.33
		Floral addition: floral density	-0.43	0.22	-1.94	0.06
E. Site fidelity ~ Aphid density	183	Intercept	3.79	0.82	4.60	< 0.001
		Cabbage aphid density	0.11	0.32	0.33	0.75

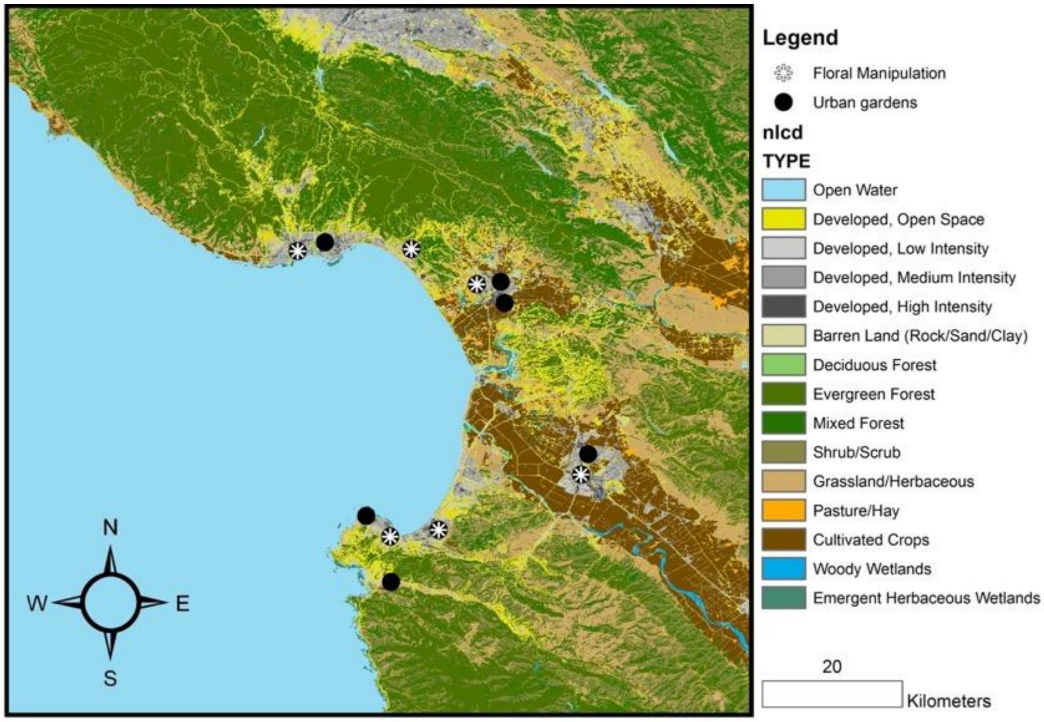

Figure 3-1. Urban gardens in the California central coast in which the two-phase study took place. Six of the 12 sites received a floral resource addition; all 12 sites received the ladybeetle population manipulation. The gardens are surrounded by differences in landscape composition (i.e., land cover classes) classified by the National Land Cover Database (Jin et al. 2013).

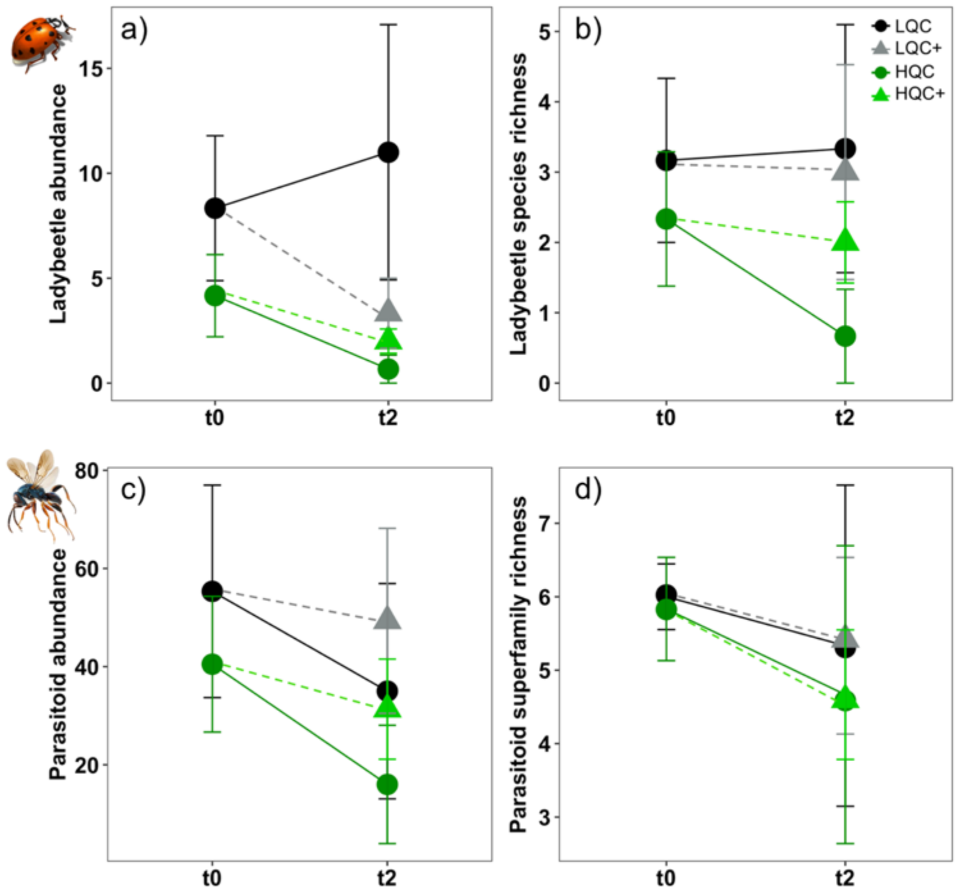

Figure 3-2. Mean ladybeetle abundance (a), ladybeetle richness (b), parasitoid abundance (c) and parasitoid superfamily richness (d) observed in the 12 gardens of different landscape types before (t_0) and after (t_2) the floral resource addition experiment. Bars show standard error of the mean with 95% confidence interval. Where HQC = high quality cityscape; LQC = low quality cityscape; and "+" represents the addition of floral resources.

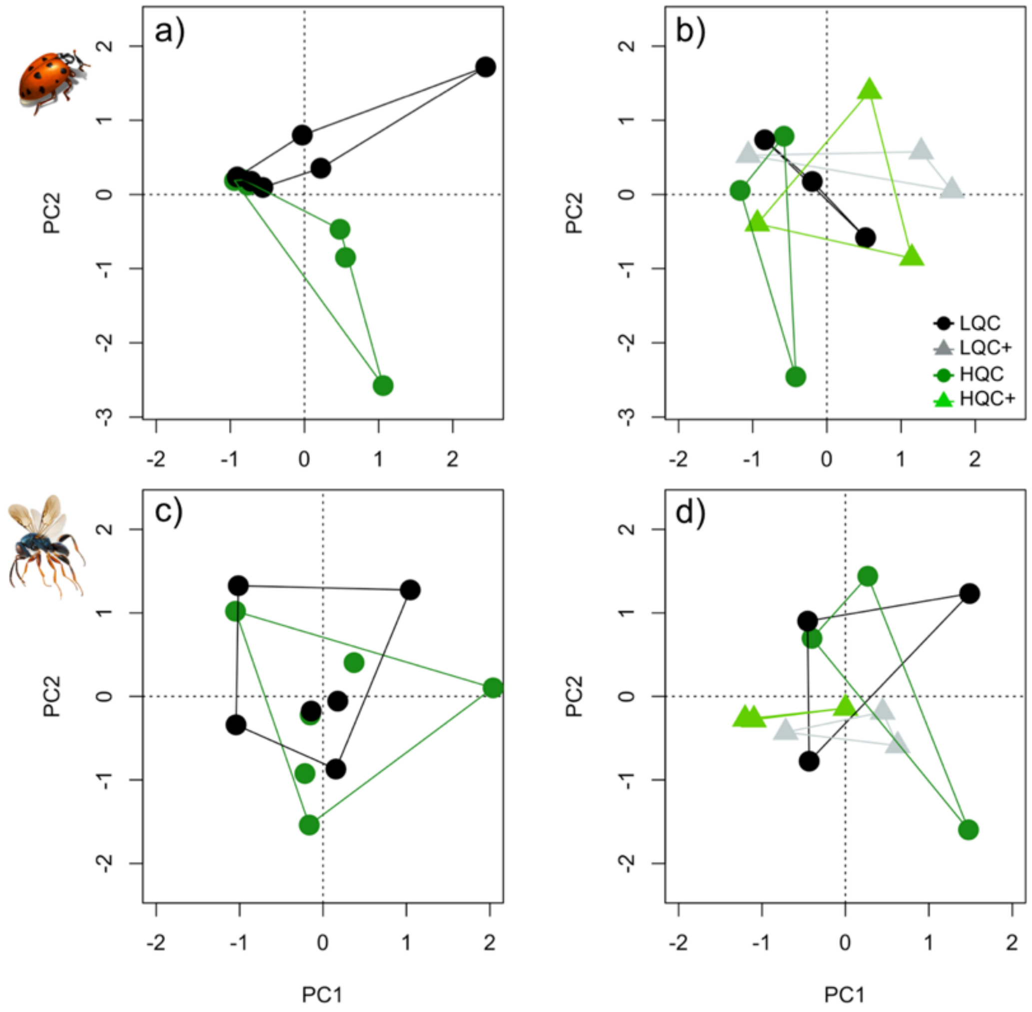

Figure 3-3. Composition of ladybeetle communities (to species) before (a) and after (b) the floral resource addition experiment in gardens surrounded by two landscape types (HQC= high quality cityscape; LQC= low quality cityscape) with (“+”) or without the floral resource addition. Composition of parasitoid wasp communities (to superfamily) before (c) and after (d) the floral resource addition experiment.

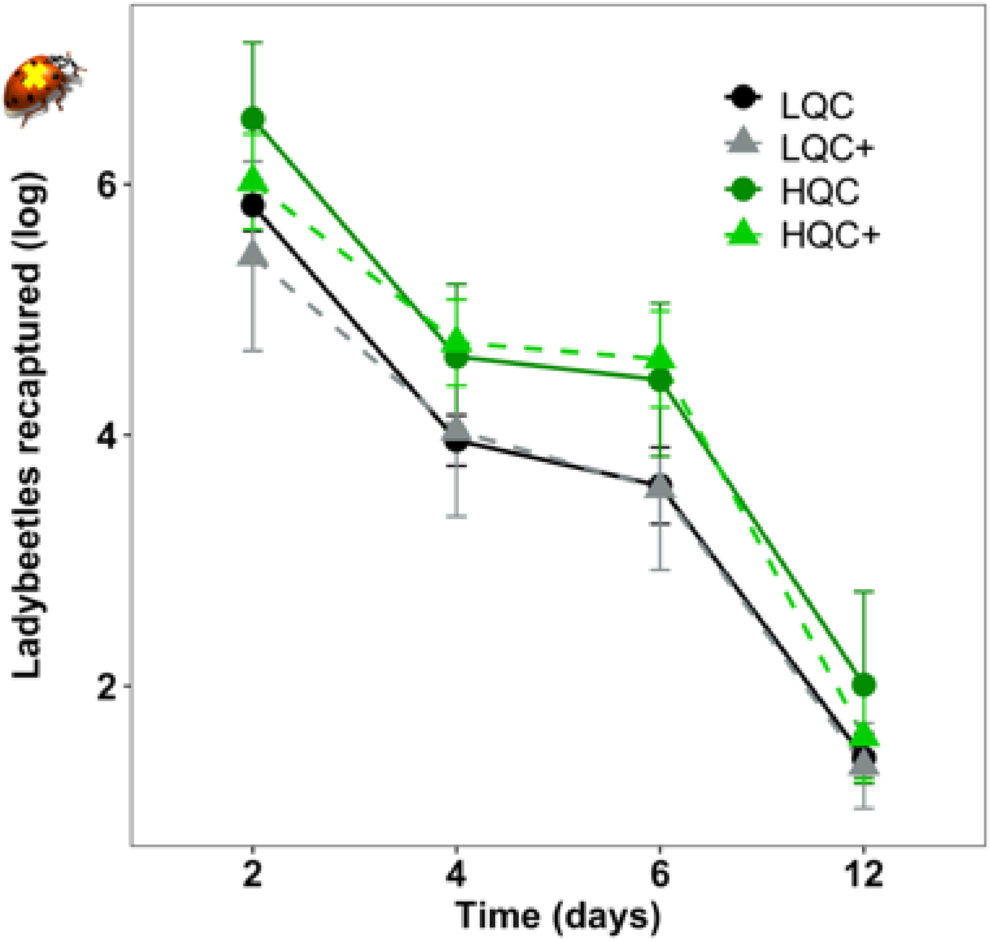

Figure 3-4. Site fidelity - the number of marked ladybeetles recaptured (log transformed) – to gardens surrounded by two landscape types (HQC = high quality cityscape; LQC = low quality cityscape) with ("+") or without the floral resource addition over the study period. Bars represent the standard error of the mean with 95% confidence interval.

4. Herbivore regulation in urban agroecosystems: direct and indirect effects

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Published in *Basic and Applied Ecology*, 2018

Abstract

Urban agroecosystems can provide habitat for biodiversity and can benefit human communities through urban food provisioning. Moreover, urban agroecosystems could be managed so as to optimize ecosystem services like natural pest control provided by trophic interactions between natural enemies and herbivores. As in other ecosystems, predation and parasitism regulate herbivores in urban settings, but less is known about the relative importance of direct and indirect effects at local and landscape scales in highly managed urban agroecosystems. We collected data on herbivore (cabbage aphid) density and parasitism ratios (proportion of parasitized aphid “mummies”) in 25 community gardens in three counties in the California central coast, USA. We used structural equation modeling to examine the effects of direct factors (host plant characteristics and parasitism) and indirect factors (soil, garden, and landscape characteristics) on herbivore density changes at two time points in the growing season (June and August). Aphid density, but not parasitism, varied across counties over the season, and there was a strong negative relationship between aphid density and parasitism. Direct effects were strong drivers of aphid density but not parasitism. In June, aphid density increased with host plant volume but decreased with greater floral density, while parasitism was only influenced by aphid density. In August, host plant volume similarly positively affected aphid density, and soil water holding capacity increased host plant volume. In addition, host plant density had a strong negative effect on parasitism. Urban gardeners may be able to reduce aphid pest densities by increasing floral resource density and strategically spatially distributing host plants throughout garden beds, though these processes depend on the season. The indirect effects of soil water holding capacity on aphid densities further suggest a critical role of human management on pest populations and pest control services through soil amendments and irrigation.

Keywords: pest control; urban agroecosystems; path analysis

Introduction

Drivers of trophic interactions among predators and their prey may be fundamentally different in urban ecosystems from natural systems due to the anthropogenic alteration of local resource availability and urban landscape structure (Shochat et al., 2010). Plant nutrient and water availability, for example, are usually heavily supplemented in cities, thus reducing resource limitation for herbivores (Raupp, Shrewsbury, & Herms, 2010). Local vegetation simplification and habitat disturbance, fragmentation, and isolation that are characteristic of urban ecosystems (Faeth, Warren, Shochat, & Marussich, 2005; Niemelä, 2011) can strongly influence organisms in higher trophic levels (Burkman & Gardiner, 2014; Marzluff, 2001) to potentially alter interactions between herbivores and plants (Nelson & Forbes, 2014).

In particular, urban agroecosystems like community (i.e., allotment) gardens offer a model system in which to examine how changes in local management factors and surrounding landscape features affect insect communities and their interactions. In contrast to other urban ecosystems like turf-grass dominated parks managed by the city or private home gardens managed by individuals, urban community gardens are unique in that they have multiple plot holders and generally have high local vegetation complexity and high diversity of management practices (Guitart, Pickering, & Byrne, 2012). Furthermore, urban agroecosystems are usually small habitats isolated from one another and from other natural habitats (Faeth et al., 2005). Moreover, ecological knowledge gained from garden studies has practical application because bans on synthetic pest control inputs (e.g., organophosphates, pyrethroids) in many community gardens require efficient natural pest control of herbivores by natural enemies for urban agricultural sustainability (Oberholtzer, Dimitri, & Pressman, 2014).

Environmental factors such as local vegetation composition and complexity, as well as the amount of impervious cover (i.e., concrete and built surfaces) in the urban landscape can directly and indirectly affect higher trophic interactions like parasitism of herbivores across spatial scales (Pereira-Peixoto, Pufal, Staab, Martins,

& Klein, 2016; Fenoglio, Werenkraut, Morales, & Salvo, 2017). For example, as predicted by the *resource concentration hypothesis* (Root, 1973), high host plant density in urban yards and parks increases the likelihood of pest outbreaks by directly increasing resource availability for herbivores (Dreistadt, Dahlsten, & Frankie, 1990; Shrewsbury & Raupp, 2006, 2010). Similarly, as predicted by the *plant stress hypothesis* (White, 1969), soil nutrient and irrigation management can indirectly affect sap-sucking herbivore populations by altering host plant quality (Hanks & Denno, 1993; Mattson, 1980). These local-level factors can also have indirect effects on herbivore populations by altering the abundance, species composition, and the foraging behavior of their natural enemies (Hanks & Denno, 1993; Shrewsbury & Raupp, 2000, 2006). For example, *sensu* the *enemies hypothesis*, natural enemies should be more abundant and herbivore regulation more effective by delivering greater pest mortality in more structurally complex habitats with, for example, diverse vegetation (Root, 1973). Indeed, parasitism in urban contexts has been shown to increase with increasing habitat complexity through greater natural enemy richness in those habitats (Fenoglio, Videla, Salvo, & Valladeres, 2013).

Likewise, landscape-level factors, such as the amount of impervious cover, may directly affect herbivores through changes to micro-climate (e.g., heat island effects) and atmospheric conditions (Newman, 2003) and habitat isolation (Turrini, Sanders, & Knop, 2016). The amount of impervious cover can also indirectly affect herbivores by altering the populations and communities of their natural enemies (Bennett & Gratton, 2012a; Burkman & Gardiner, 2014) who may differ in sensitivity to urbanization (Fenoglio, Salvo, & Estallo, 2009; Fenoglio, et al., 2013). For example, landscape-level environmental factors may subsequently change natural enemy-herbivore interactions (Shrewsbury & Raupp, 2000) and natural pest control provided by natural enemies through direct predation (Philpott & Bichier, 2017) and parasitism (Pereira-Peixoto et al., 2016). In sum, herbivore populations in urban community gardens can be affected by local factors through direct effects (e.g., by changing the availability and quality of their food) or through indirect ones (e.g., by changing the abundance of their natural enemy), as well as by landscape factors through direct effects (e.g., impervious cover can hinder their colonization to and from suitable habitats) or indirect ones (e.g., impervious cover can act as a colonization barrier to their natural enemies).

The strength of these direct and indirect effects may experience seasonal changes. The abundance and diversity of arthropod natural enemies can decrease over time in response to fluctuations in precipitation and temperatures (Bolger et al., 2000), and this can affect herbivore regulation through resource availability (Faeth et al., 2005). Moreover, the seasonal fluctuations that affect population-level resource availability and environmental stressors are combined with direct effects from human activities in cities that alter resources (Faeth et al., 2005). The altered patterns in resources and stressors due to temporal change and anthropogenic change can impact ecological predictions organized around direct versus indirect effects, resource concentration versus natural enemy regulation, at local versus landscape scales in urban systems (Dale & Frank, 2014). Thus even though we are beginning to understand the local and landscape factors that regulate herbivores through parasitism in urban systems (Fenoglio et al., 2013; Pereira-Peixoto et al., 2016), we still lack an understanding of local, landscape, and temporal factors in urban agroecosystems compared to rural agricultural landscapes.

In this study we aimed to investigate how local and landscape factors directly and indirectly affect insect herbivore regulation in urban agroecosystems (community gardens). We focus on the regulation of aphid herbivores – a prevalent sap-sucking pest in gardens – through parasitism by parasitic wasps as a trophic interaction that provides pest control services. Specifically, we asked: 1) How do local garden management factors (host plant characteristics, floral resources, and soil properties) and the landscape context of gardens (amount of surrounding impervious cover) affect herbivore abundance through direct effects on herbivore density and indirect effects by affecting herbivore parasitism? 2) Does the importance of these effects change with time?

Materials and methods

Study system

We worked in 25 community gardens in three counties in the California central coast, USA: Monterey (36.2400° N, 121.3100° W), Santa Clara (37.3600° N, 121.9700° W), and Santa Cruz (37.0300° N, 122.0100° W) (Fig. 1). The gardens range from 405 to 15,525 m^2 in size, are separated from one another by > 2 km, and are surrounded by a mix of natural, agricultural, open green space, and impervious land cover (Fig. 1).

Aphid populations and parasitism

Due to its widespread abundance, its economic importance, and the presence of *Brassica* plants in all of our gardens, we chose cabbage aphids (*Brevicoryne brassicae*) as our model organism to examine local and landscape factors that affect herbivore regulation. Cabbage aphids are the most common and abundant aphid species on *Brassica* crops (e.g., cabbages, Brussels sprouts, kale) and, in urban agroecosystems, cabbage aphids infest crops and shorten crop life (Flint 2013). While other aphid species are present in the gardens (e.g., potato aphids (*Macrosiphum euphorbiae*), bean aphids (*Aphis fabae*), green peach aphids (*Myzus persicae*)), cabbage aphids made up 99% of the individuals counted, and thus we decided to only focus on cabbage aphids for our study. Cabbage aphids are consumed by arthropod predators like ladybeetles and attacked by parasitoid wasps (Hymenoptera). Although ant-tending can deter predators and parasitoids (Müller & Godfray, 1999), in our study sites fewer than 2% of *Brassica* plants with cabbage aphids are tended by ants (Philpott S., unpublished data). While we have previously measured predation by other arthropods (e.g., spiders, ladybeetles, wasps) in this system (Philpott & Bichier, 2017), we have yet to measure the effect of parasitism on aphid regulation. Thus in this study we focused on parasitism of aphids.

The study gardens have abundant and diverse parasitoid wasp communities (Burks & Philpott, 2017). Parasitoids can provide significant pest control in urban ecosystems (Burkman & Gardiner, 2014), and are considered among the most important natural enemies of aphids in rural and urban agroecosystems (UC IPM, 2009, 2013). Parasitized aphids (i.e., "mummies") can be easily recognized by their leathery brown "bloated" bodies, and therefore easily quantified.

During the summer of 2016, we did monthly counts of cabbage aphid abundance and parasitism rates on *Brassica* plants in each garden (May 24 - June 7, June 27 - 30, July 25 - 28, August 22 - 26). We visually surveyed cabbage aphids and aphid parasitism on five randomly chosen *Brassica* plants in five haphazard locations within a 20 x 20 m survey plot at the center of each garden. Different plants were sampled on the different sampling occasions. For each plant, we counted the number of leaves, measured the plant height, and counted the number of non-parasitized and parasitized cabbage aphids. We used the University of California Agriculture and Natural Resources Integrated Pest Management Program's Guidelines to identify cabbage aphid individuals and colonies (UC IPM, 2009). We collected parasitized aphid mummies (n = 180) and reared them in the laboratory in plastic vials with a cotton stopper to identify parasitoid species for a qualitative assessment of natural enemies present. Parasitoids were identified to family using Naumann (1991). To measure parasitism rates we calculated a parasitism ratio following Roschewitz et al. (2005) (# mummies / (# aphids + # mummies)) and calculated host plant volume (number of leaves x plant height) for all *Brassica*.

Local and landscape factors

We collected data on garden environmental factors that we hypothesized directly and indirectly regulate aphid populations because they affect parasitoids in these gardens (Burks & Philpott, 2017; further described below). At two time points while sampling for aphids (late June and August 2016), we counted the number of *Brassica* plants within the 20 x 20 m plots for a measure of host plant density, and counted the number of flowers within eight random 1 x 1 m sub-plots within the 20 x 20 m plots because flowers are important nectar resources for parasitoids in agroecosystems (Balzan & Wäckers, 2013). We sampled soils for inorganic nitrogen content because increased nitrogen (N) increases plant phloem quality thereby potentially increasing the abundance of sap-sucking herbivores, such as aphids (Mattson, 1980; Nowak & Komor, 2010). We sampled soils rather than plant N because we did not have permission from all gardeners within the community gardens to remove plant material. We use soils as an indicator of plant N with recognition that these two metrics are not the same, but that plant N is often correlated with soil N in agroecosystems (Hofman & Van Cleemput, 2004). We sampled soils at the same five *Brassica* plants as the aphid surveys, taking three 0-20 cm soil cores within 10 cm of the plant's base with steel soil augers. We aggregated soil samples for one representative sample for each plant and performed a KCl extraction with fresh soil within 24 h following Wilke (2005). In addition, we sampled soils to determine soil water holding capacity (WHC), using a cylindrical metal sampler at four 1 x 1 m sub-plots per garden. Soil WHC is an indicator of soil fertility due to its influence on crop growth ability and is a standardized measure for indicating long-term water availability because it is less sensitive to sampling instance (i.e., when gardeners watered). Following Wilke (2005), we screened soils through a 2 mm sieve, filled a 2 x 2’’ cylinder with a perforated base with field-moist soil, and submerged cylinders in a water bath for 8 h. We removed, capped cylinders (to avoid evaporation), and placed cylinders on a tray of sand. Once a consistent weight of the soil sample was achieved over time, we dried the soil at 105 °C for 24 h.

We measured garden size (m^2) using Google Earth imagery (Google, 2016) because changes in habitat size may affect natural enemies and aphids in urban landscapes (Gibb & Hochuli, 2002). We used land cover data from the US Geological Survey’s National Land Cover Database (NLCD) (Jin et al., 2013) to measure the proportion of urban developed land cover (determined by the amount of impervious or built cover) within buffers at a 1 km spatial scale surrounding each garden. We chose to focus on urban developed land because it may affect aphids and natural enemies by acting as a barrier to dispersal, and we selected 1 km because aphids and parasitoids respond to landscape factors at this spatial scale both in rural (Gagic et al., 2011; Roschewitz, Hücker, Tscharntke, & Thies, 2005) and urban (Egerer et al., 2017a; Nelson & Forbes, 2014) landscapes. Using spatial statistics tools in ArcGIS (v 10.1) (ESRI, 2011), zonal histograms identified the total proportion cover of three NLCD land cover classes present within each buffer that represent urban land cover (Homer et al., 2015): 1) low density housing land cover (49% impervious cover; land cover class # 22), 2) medium density housing land cover (50-79%; # 23), and 3) high density housing and buildings (80-100%; # 24). We combined the three classes for a total proportion of urban land cover surrounding gardens at 1 km. Here, a high total proportion of urban land cover indicates higher degrees of urbanization, and a low proportion indicates low degrees of urbanization.

Analysis

We used generalized linear models to compare aphid density and parasitism ratios among counties and across sampling period. The number of aphids observed and the calculated parasitism ratio per *Brassica* survey were the response variables, and county and sampling period were the predictors. The aphid density model was fit with a Poisson distribution and the parasitism model was fit with a negative binomial distribution. We fit the full model and ran a post-hoc test using the *glht* function in the *multcomp* package (Hothorn, Bretz, & Westfall, 2008) in the R statistical environment (R Development Core Team, 2016) to assess differences between counties and sampling periods. Second, we constructed a conceptual path analysis model, a form of structural equation modeling (SEM) (Mitchell, 2001), to understand the relative importance of direct and indirect effects on aphid density using a priori knowledge of the local and landscape factors that affect parasitoid abundance, species richness, and species composition in the same gardens (see Burks & Philpott, 2017) (Fig. 2). In the model, we predicted that host plant size and density (*sensu* the *resource concentration hypothesis*), and proportion of urban land cover would positively directly affect herbivore density (Newman, 2003; Turrini, Sanders, &

Knop, 2016). We also predicted that soil properties (WHC, inorganic nitrogen (NO_3-) concentration) would positively indirectly affect herbivore density by directly affecting plant size and quality (*sensu* the *plant stress hypothesis*). Moreover, we predicted that garden size and floral abundance (density) would positively directly affect parasitism (*sensu* the *enemies hypothesis*), while the proportion of urban land cover would negatively directly affect parasitism to indirectly affect herbivore density through a connection between parasitism and herbivore density (i.e., via trophic cascade). Last, we predicted that effects would change over the season due to fluctuations in populations and thus trophic interactions throughout the growing season in response to changes in environmental factors and agricultural management.

We built the SEM with generalized linear mixed effects models using piecewiseSEM (Lefcheck, 2016) in R. This approach (1) accounted for the hierarchical structure of the data (*Brassica* nested within site, nested within counties), (2) allowed for the incorporation of random effects (site nested within county) and (3) accepted flexible response variable distributions. For each mixed effects component model, we added a random effect. The aphid density component model was fit with a Poisson distribution, the parasitism model was fit with a negative binomial distribution, and the host plant volume model was fit with a Gaussian distribution. Explanatory variables were the averaged local factors measured at each respective sampling period for each site with the exception of soil WHC, proportion of urban land cover, and garden size, which we assumed did not change significantly across the season and thus were collected once per year. Explanatory variables were standardized and centered prior to analysis. To examine whether patterns change over time, we built two SEMs to analyze early growing season pathways (late June) and later growing season pathways (August) (see Table 1 for how variables differ by month). For each model we obtained standardized regression coefficients using the *sem.coefs* function and model fit tests using the *sem.model.fits* function. Here, good model fits are determined by a Fisher's C statistic calculated by the significance of all missing paths (Shipley, 2009), a X^2 test (the model is an adequate fit when $P > 0.05$), and Akaike's information criterion (AICc) value (Lefcheck, 2016).

Results

Trends in aphid density and parasitism

Cabbage aphid density, but not parasitism, on *Brassica* significantly varied in gardens across counties and sampling periods (Fig. 3; Table 2). Santa Cruz gardens had significantly lower aphid densities than Monterey ($P < 0.001$) but greater aphid densities than Santa Clara ($P = 0.01$), whereas Monterey and Santa Clara gardens significantly differed from one another ($P < 0.001$; Fig. 3A; Table 2A). Cabbage aphid densities significantly differed in late June ($P < 0.001$) and July ($P < 0.001$) from May, but not in August ($P = 0.147$; Table 2A). Parasitism on *Brassica* generally declined from May to July across sites among counties, but then increased in August (Fig. 3B; Table 2B). Mean parasitism per site ranged from 0% parasitism in May to up to 26% in August (Table 1). Of the 180 reared aphid mummies, 33 parasitoids emerged (18.3%), consistent with low rearing success in other studies (van Veen, Morris, & Godfray, 2006). The identified parasitoid families that emerged from collected cabbage aphid mummies were Pteromalidae (36.4%; n = 12 individuals), Braconidae (30.3%; n = 10), and Cynipidae (27.3%; n = 9). To note, Braconidae species are primary aphid parasitoids, ovipositing directly inside of the aphid that leads to permanent aphid paralysis and mummification (Stary, 1970). Many species of Pteromalidae (Superfamily: Chalcidoidea) and Cynipidae (Cynipoidea) are secondary parasitoids, or hyperparasitoids, that attack aphids but delay development until aphid mummification by the primary parasitoid occurs (Müller, Adriaanse, Belshaw & Godfray, 1999).

Direct and indirect effects

The conceptual SEM had predictive power of the direct and indirect effects of local and landscape factors on aphid density and parasitism across the season (June: Fisher's C_{12} = 14.13, P = 0.29, AICc = 81.26; August C_{12} = 6.14, P = 0.91, AICc = 70.27) with a higher model fit in August. Cabbage aphid density was significantly negatively correlated to parasitism over the season, both in June and August (coefficients listed in Fig. 4, see Appendix A: Table 1; Fig. 1). Local factors directly affected cabbage aphid density but their importance varied between seasonal time points. Floral density significantly negatively affected June cabbage aphid densities (P = 0.005), host plant density positively affected August aphid densities (P = 0.009), and host plant volume had a significantly positive effect on aphid densities in both June (P < 0.001) and August (P = 0.004) (Fig. 4A; see Appendix A: Table 1). Soil water holding capacity had a significant positive effect on host plant volume in August (P = 0.01), indirectly affecting aphid densities (Fig. 4B). Local factors only indirectly affected parasitism across the season through the connection between aphid density and parasitism. The landscape factor (urban land cover) did not have significant effects on either aphid density or parasitism.

Discussion

We found strong evidence of local garden management factors directly and indirectly affecting herbivore populations. We also found a strong negative correlation between parasitism ratios and cabbage aphid population density. Contrary to expectations, we did not find any significant effects of the amount of urban land cover in the landscape on aphid densities nor on parasitism. Furthermore, none of the predicted factors had any direct effects on parasitism. The relative importance of the effect of management varied temporally as indicated by a higher fit of the August model, and the significant differences in aphid densities over time. Consequently, we focus on interpreting the August model in the following discussion.

Influence of garden management factors on herbivores

A soil management factor, water holding capacity (WHC), directly increased host plant volume, to have positive indirect effects on aphid densities. High soil WHC is associated with more soil organic matter and improved soil fertility through soil and groundcover amendments in these gardens (Egerer et al., 2017b). Such management practices (e.g., composting) have been shown to reduce plant water stress and enhance plant growth, which can then lead to aphid population growth (Archer, Bynum, Onken, & Wendt, 1995). Garden soils with higher WHC retain moisture for longer to prevent water stress – especially later in the season – and thereby can support host plant growth to ultimately support more herbivores. These findings follow predictions that host plant condition matters for herbivores, in particular that larger host plants can support higher aphid densities (Dixon, 1977).

Our results suggest that nitrogen availability in the soil may not be as important as plant water content for aphid populations in these gardens. The fact that plant turgor pressure and cell water content is necessary for phloem feeders' nitrogen utilization (Huberty & Denno, 2004) could explain why we found no direct effect of soil inorganic nitrogen on host plant volume or no indirect effect on aphid density. Our findings could also mean that either this measure of plant condition was too coarse, that plant quality may be relatively similar in gardens (particularly in relation to outside garden vegetation), or that fertilization may not universally increase herbivore population loads in urban systems (Dyer & Stireman, 2003).

In accordance to the *resource concentration hypothesis* (Root, 1973), we found that greater host plant densities resulted in greater aphid densities in the gardens. These results concur with studies in rural agricultural systems (Andow, 1990), in urban home gardens (Shrewsbury & Raupp, 2006), and other green space patches (Fenoglio et al., 2009). Greater *Brassica* density across the garden habitat provided an abundant resource for aphid populations to exploit, as well as shorter travel distance between host plants. Decreasing proximity between plants, often associated with higher plant density, likely facilitates herbivore movement from and colonization to new host plants (Hambäck, Björkman, Rämert, & Hopkins, 2009) and is particularly important for herbivores like aphids that reproduce quickly and disperse with increasing colony density on plants (Dixon, 1977). Since increasing herbivore density increases the likelihood of plant damage (Kim & Underwood, 2015), it would be valuable to further examine the maximum density of conspecific plants in a garden before increases in damaging herbivore populations occurs.

Influence of garden management factors on herbivore-parasitoid interactions

The strong negative relationship between aphid density and parasitism suggests that parasitoids can regulate herbivore populations through top-down pathways. Although parasitism was not as high at high aphid densities, even ~ 30% parasitism may be sufficient to keep aphid densities from increasing further. Surprisingly, none of the local or landscape drivers had direct effects on parasitism, meaning that parasitism is driven by the availability of herbivore hosts but not necessarily garden management factors. In particular, the number of flowers within gardens directly negatively affected aphid densities, but counter to our predictions did not directly affect parasitism. This contrasts with previous studies of parasitism in rural agricultural systems where parasitism increases with agroecosystem floral availability (Jonsson, Wratten, Landis, Tompkins, & Cullen, 2010), but supports other findings that local factors including floral availability have no direct effect on parasitism in urban systems (Hanks & Denno, 1993; Dale & Frank, 2014;

Lowenstein, Gharehaghaji, & Wise, 2017). Flowers may, however, instead attract other aphidophagous predators that are negatively affecting aphid densities. Indeed, ornamental flowers increase ladybeetle abundance in our system (Egerer, Bichier, & Philpott, 2017), and syrphids, whose larvae predate aphids, are generally supported by floral resources in agroecosystems (Haenke, Scheid, Schaefer, Tscharntke, & Thies, 2009; Landis, Wratten, & Gurr, 2000). While here we focus on the role of parasitism for herbivore regulation, we acknowledge that other predators could reduce aphid densities and influence parasitism ratios by altering the density and spatial distribution of prey, or by consuming parasitized individuals to skew observations. Aphid-tending ants could potentially affect aphid densities (Müller & Godfrey, 1999), but the few observations (2%) of *Brassica* with ants in these gardens (as described above) suggest that ants are not playing a strong role in inhibiting *Brassica* parasitoids in these gardens.

Other local vegetation characteristics not measured here may be important drivers of parasitism. A recent study found that not only host plant species diversity but greater plant phylogenetic diversity may be important for decreasing herbivore densities and increasing parasitism by providing more microhabitats and microclimates within habitats to support parasitoid communities (Staab et al., 2016). Therefore, incorporating plant structural diversity (e.g., reducing concentrations of host plants in plots) and phylogenetic diversity (e.g., increasing the number of cultivated varieties of host plants) in urban gardens may be important for increasing parasitoid abundance and diversity to thereby increase parasitism.

Influence of urban context on herbivore-parasitoid interactions

The proportion of urban cover in the landscape did not directly or indirectly affect aphid densities or parasitism in the studied community gardens. Herbivores like aphids may be less susceptible to landscape-scale urban environmental disturbances due to relatively high local host plant abundance and quality within gardens compared to the surrounding environment. Here, gardens provide an irrigated and productive habitat with more resources during drought conditions (Faeth et al., 2005) like in California during this study, and could influence aphid densities. Aphids can also disperse relatively long distances to locate habitat patches as they are carried by wind currents (Compton, 2002), and this passive long-distance dispersal ability may be why aphid densities are unaffected by changes in the amounts of impervious cover in the landscape. Although parasitoids are generally more sensitive to land use intensification (Tscharntke & Brandl, 2004) like urbanization (Denys & Schmidt, 1998), our findings are comparable to other urban studies that found no strong effects of urbanization on variation in herbivore parasitism (Bennett & Gratton, 2012b; Fenoglio et al., 2009; Lowenstein et al., 2017). This previous work suggests that some parasitoid species may be more tolerant of anthropogenic disturbance, particularly those that are efficient natural enemies, to maintain similar levels of pest control across gradients in landscape-scale urbanization (Fenoglio et al., 2009).

Temporal variation in herbivore regulation, herbivore-parasitoid interactions

The relative importance of direct and indirect local effects changes over the season, as indicated by the difference in model fit between months and the relative effect of local variables. This may be attributed to the observed fluctuations and variability in aphid densities across sampling periods across counties. Yet, while aphid densities fluctuated across the season, across counties, and with changing management factors, parasitism did not differ over time despite these fluctuations in their hosts. This suggests that although parasitoid communities are linked to aphid densities, they are likely limited in their ability to regulate herbivores at high densities. The differences in responses over the season could further be explained by abiotic factors that we did not directly measure, such as increasing urban temperatures that can boost herbivore fitness on host plants without affecting herbivore regulation by natural enemies (Dale & Hanks, 2014).

Conclusions

Population dynamics and trophic interactions are modified in urban agroecosystems through human land management at local and landscape scales. In community gardens, aphid herbivore pest population regulation can be driven by direct effects (through vegetation properties) and indirect effects (through soil properties) on herbivores predominantly through bottom-up pathways. Moreover, local management of soil and vegetation within gardens was here more important for aphid pest control than the degree of urbanization surrounding gardens. This suggests that gardeners can, to some degree, regulate aphid populations through vegetation and soil management. Vegetation structure and composition, particularly of *Brassica* crops that are favorites of both gardeners and aphid herbivores, may be strategically planned or intercropped within garden beds and across the garden habitat to lower aphid densities. Further, interspersing flowering plants or diversifying the crops planted may reduce aphid densities as well. Changes in soil management, however, may not benefit gardeners due to important agroecosystems trade-offs. On the one hand, building soil organic matter through composting or mulching may increase soil water holding capacity, boost plant growth, and conserve water in urban agroecosystems (Edmondson, Davies, Gaston, & Leake, 2014), but our findings suggest that related increases in plant quality may boost aphid populations, without increases in parasitism. In order to maintain high plant quality, while also managing pests, gardeners should carefully monitor herbivore populations, and follow other general recommendations for supporting parasitoid and natural enemy populations within gardens (Raupp et al., 2001; Shrewsbury, Lashomb, James, Patts, & Casagrande, 2004; Flint, 2013). In sum, agroecosystem management should recognize and balance the trade-offs that result from the concert of direct and indirect effects in agroecosystem pathways to ultimately enhance pest control services.

Table 4-1. Overview of the minimum, maximum, mean (± standardized error of the mean) values of the local management factors averaged per site across all 25 garden sites at the two time periods measured.

	June			**August**		
	Min	**Max**	**Mean ± SE**	**Min**	**Max**	**Mean ± SE**
Aphid density (avg. plant)	14	82	42 ± 3	1	65	31 ± 3
Parasitism (ratio)	0.04	0.50	0.24 ± 0.02	0.20	0.82	0.37 ± 0.03
Floral density (1 x 1 m)	1	1,328	192 ± 56	1	470	48 ± 21
Brassica density (20 x 20 m)	2	371	55 ± 16	4	262	41 ± 12
Host plant volume[a]	4.3	9.2	6.7 ± 0.1	4.5	8.5	6.6 ± 0.1
Soil inorganic N (mg/kg soil)	0.1	8.2	1.5 ± 0.4	0.1	13.2	2.4 ± 0.6

[a] Average plant height multiplied by the number of leaves, natural log transformed.

Table 4-2. Analysis of (A) cabbage aphid density and (B) parasitism using generalized linear regression models. Parameters indicate the counties in which the community gardens are located (Santa Cruz county as reference level), and the sampling period (month) (May as reference level). Significant differences among counties and months assessed through post-hoc comparisons indicated with superscripts.

	Parameter	**Coefficient**	**SE**	**z**	**P**
A. Aphid density	**County**				
	Intercept (Santa Cruz)[a]	3.54	0.01	246.47	< 0.001
	Santa Clara[b]	-0.06	0.02	-2.91	0.004
	Monterey[c]	0.14	0.02	7.70	< 0.001
	Sampling period				
	Intercept (May)[a]	3.38	0.02	202.27	< 0.001
	June[b]	0.36	0.02	16.54	< 0.001
	July[b]	0.35	0.02	15.93	< 0.001
	August[a]	0.03	0.02	1.45	0.147
B. Parasitism	**County**				
	Intercept (Santa Cruz)[a]	-0.82	0.15	-7.76	< 0.001
	Santa Clara[a]	0.04	0.21	0.17	0.865
	Monterey[a]	-0.15	0.21	-0.71	0.481
	Sampling period				
	Intercept (May)[a]	-0.82	0.16	-7.20	< 0.001
	June[a]	-0.23	0.25	-0.94	0.345
	July[a]	-0.20	0.24	-0.82	0.410

August[a]	0.18	0.22	0.81	0.419

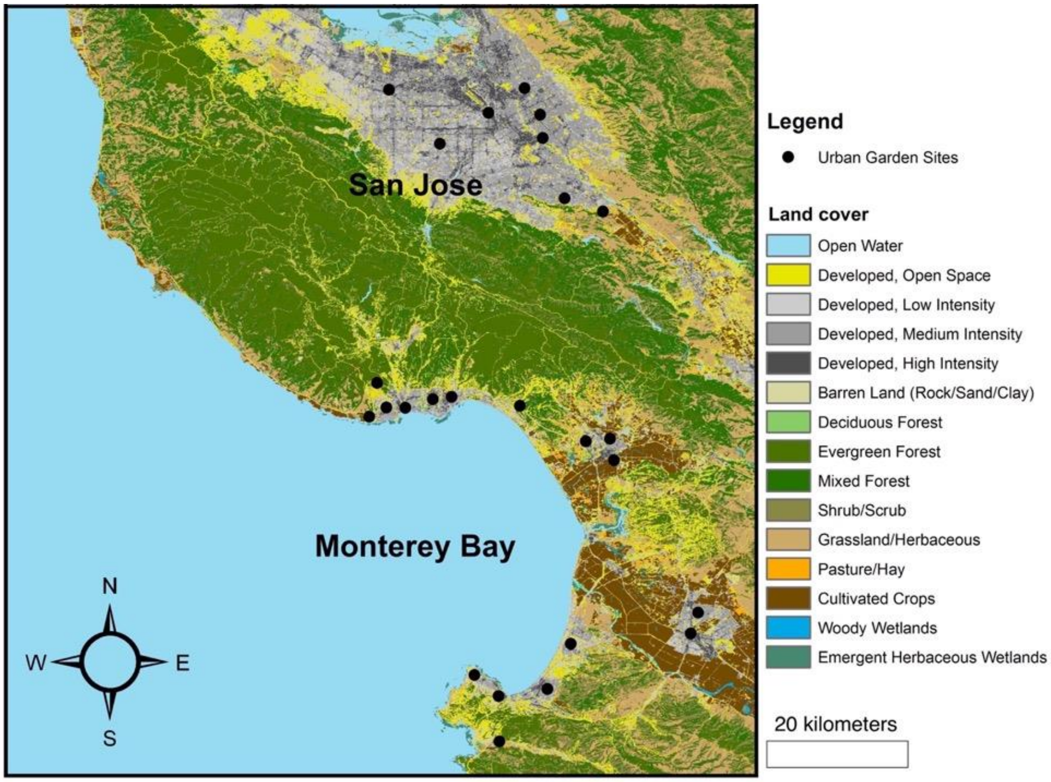

Figure 4-1. Map of the California central coast, USA showing the 25 urban community garden sites in Monterey, Santa Cruz, and Santa Clara counties. The gardens are surrounded by different land cover types as classified by the US Geological Survey's National Land Cover Database.

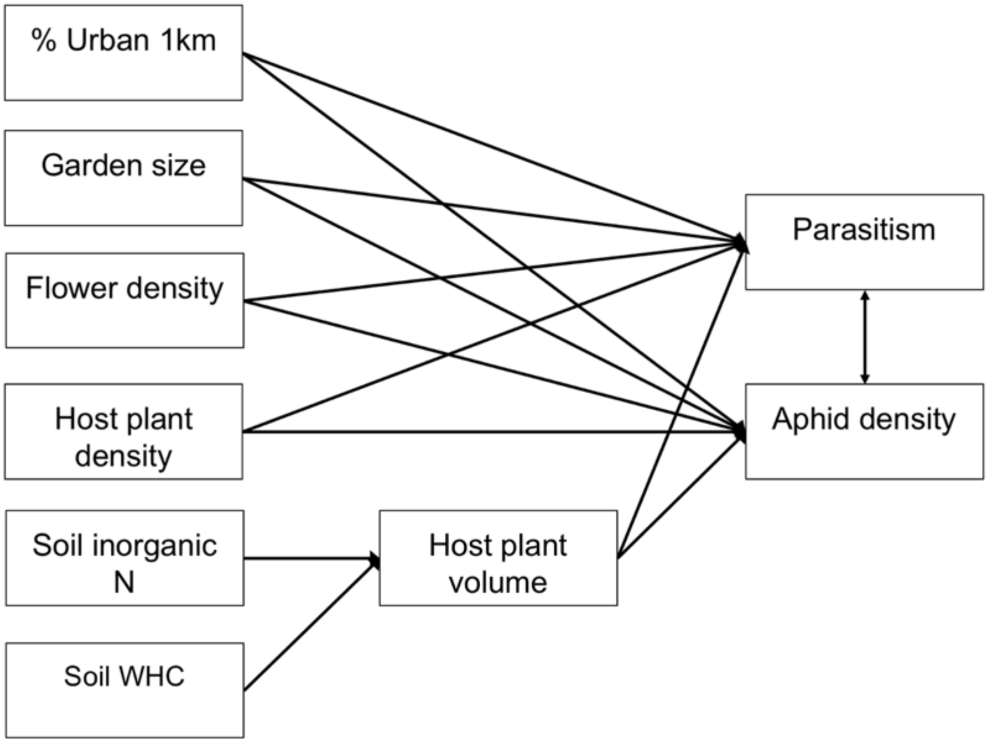

Figure 4-2. Conceptual path analysis model of direct and indirect effects on aphid density and parasitism.

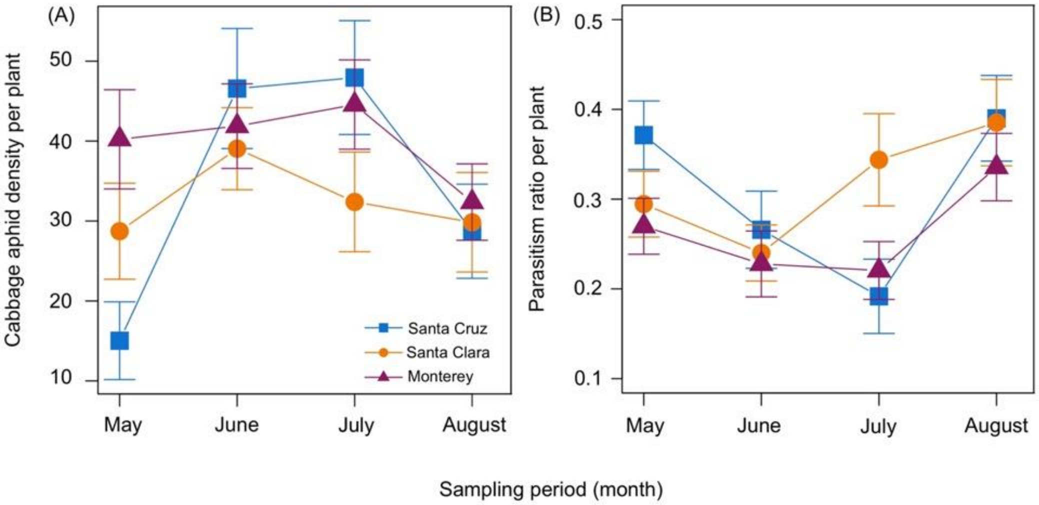

Figure 4-3. Trends in (A) cabbage aphid density and (B) cabbage aphid parasitism (ratio) across counties and across sampling periods. Symbols in the plots represent the mean value (density, parasitism) for the gardens for each county. Error bars represent standard errors (SE) of the means.

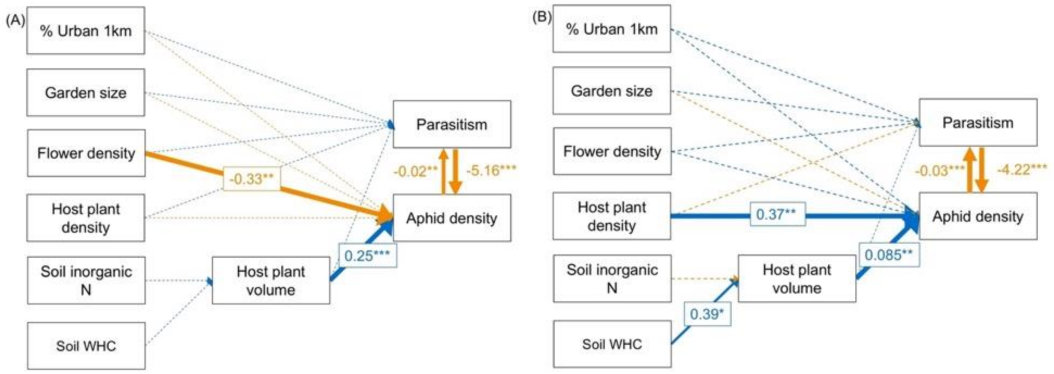

Figure 4-4. Path diagrams of garden trophic interactions for (A) June and (B) August displaying standardized coefficients of effects. Line type (solid vs dashed) indicates statistically significant effects (solid lines; * = P < 0.05, ** = P < 0.01, *** = P < 0.001; dashed lines insignificant). Blue lines indicate positive effects; orange lines indicate negative effects.

5. Context matters: contrasting ladybird beetle responses to urban environments across two US regions

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Published in *Sustainability*, 2018

Abstract

Urban agroecosystems offer an opportunity to investigate the diversity and distribution of organisms that are conserved in city landscapes. This information is not only important for conservation efforts, but also has important implications for sustainable agricultural practices. Associated biodiversity can provide ecosystem services like pollination and pest control, but because organisms may respond differently to the unique environmental filters of specific urban landscapes, it is valuable to compare regions that have different abiotic conditions and urbanization histories. In this study, we compared the abundance and diversity of ladybird beetles within urban gardens in California and Michigan, USA. We asked what species are shared, and what species are unique to urban regions. Moreover, we asked how beetle diversity is influenced by the amount and rate of urbanization surrounding sampled urban gardens. We found that the abundance and diversity of beetles, particularly of unique species, respond in opposite directions to urbanization: ladybirds increased with urbanization in California, but decreased with urbanization in Michigan. We propose that in California water availability in gardens and the urbanization history of the landscape could explain the divergent pattern. Thus, urban context is likely a key contributor to biodiversity within habitats and an important consideration for sustainable agricultural practices in urban agroecosystems.

Keywords: urban gardens; biological control; impervious surface; urbanization rate; Michigan; California

1. Introduction

Urbanization is changing biodiversity patterns and population distributions in cities across the world [1,2]. Urban environments are characterized by changes in abiotic [3] as well as biotic conditions [4]. For example, greater amounts of impervious surface in cities causes urban heat island effects, which increases the temperatures of cities [5] and within urban green spaces [6]. Light pollution from buildings and car traffic extends the duration and intensity of light availability, affecting the circadian rhythms of biotic elements [7]. Irrigation of lawns, parks and gardens adds water resources and maintains the presence of vegetation for organisms to exploit [8,9]. Moreover, the abundance and distribution of species and resources (habitat/food/shelter) in urban areas are often supplemented or altered across time and space [1,10].

Changes in environmental conditions and resource availability have different effects on different taxa and species [1,11]. Some species are able to persist and thrive in altered urban environments, while environmental filters and competition can cause other species to decline [12]. The species that thrive, what some consider "urban exploiters", are often habitat generalists that are able to live, exploit resources and reproduce in diverse, resource poor environmental conditions [2]. On the other hand, specialist species with particular habitat (food, shelter) requirements may be more sensitive to—and decline with—increasing urbanization because cities do not have the vegetation or resources to support these species [13]. The negative effect(s) of urbanization on species life history and functional traits may lead to biotic homogenization and declines in species richness within urban habitats [14].

The rate at which urbanization occurs (i.e., the speed at which land is converted to impervious surface) could further affect the diversity and distribution of species abundance and richness within urban habitats, and their ability to adapt to certain urban conditions. The percent impervious surface is forecasted to increase by 1.5 million km2 by 2030 [15,16]. Moreover, because cities have distinct development histories, socio-cultural and demographic trends [17], it is important to understand whether and how biodiversity will respond to increasing urbanization (and associated qualitative and quantitative aspects) across multiple urban environmental contexts [16,18]. Elucidating whether certain organisms respond differently or not between unique regions can inform conservation agendas and urban sustainability broadly for various cities across the world [1,19].

Urban agroecosystems such as community and home gardens are high-quality habitats that conserve considerable amounts of biodiversity in cities [20]. These systems are heavily managed by people predominantly for the purpose of cultivating fresh vegetables, fruits, flowers and herbs for self-consumption [21,22,23]. Because urban agroecosystems are usually vegetated and irrigated [24], they provide food and shelter for many arthropods. Certain arthropod groups, for example pollinators and natural enemies, are in turn important for providing ecosystem services like crop pollination and pest control. Previous studies have shown that these arthropod groups are less abundant in gardens where surrounding levels of urbanization are high [25,26]. However, groups respond differentially to urbanization and at different spatial scales [27,28]. Some arthropod groups and species in urban environments including urban agroecosystems may actually increase with urbanization [29,30]. For example, insect pollinator species diversity is greater in some urban regions compared to surrounding suburban and agricultural areas [30,31,32]. Currently, it is unclear whether these patterns are regional phenomena or if these trends are generalizable to other urban regional contexts. We argue that this question warrants further investigation, requiring research that draws comparisons of arthropod biodiversity across spatially distinct regions. Yet studies in urban agroecosystems that compare and synthesize findings across regions with different environmental conditions are rare [33].

Here, we combine data on ladybird beetle abundance and species richness collected from comparable urban agroecosystems in two distinct geographical regions to test whether the response of ladybird beetles to urbanization differs by the environmental context and urbanization history. Ladybird beetles are charismatic arthropods in agroecosystems that provide key natural pest control services, particularly of herbivorous aphids, mites and scale insects [34,35,36]. Because urban agroecosystems are situated among dense human populations, they tend towards organic, environmentally friendly, and human-health-conscious forms of management [37]. Thus, natural pest control is particularly important for these agroecosystems. In this study, we asked: (1) Does the relationship between urbanization (percent impervious surface, rate of development) and predator (ladybird beetle) systems in urban agroecosystems change with environmental context (region)? (2) Which species are shared by, and which are unique to urban agroecosystems of different regions? (3) Do shared species respond differently to urbanization measures in the region than unique species?

2. Methods

2.1. Study Regions

We worked in two regions in the USA—California and Michigan—to collect ladybird beetle data in urban community gardens in these regions (Figure 1). In California, we collected ladybird beetle data from 18 urban gardens in the California central coast in Santa Clara, Santa Cruz, and Monterey counties, which have estimated population densities of 274, 232, and 50 people/sq. km, respectively (2010–2014 U.S. census period) [38]. In Michigan, we collected ladybird beetle data from 13 urban gardens in Washtenaw county, which has an estimated population density of 192 people/sq. km [38]. The gardens in both regions are surrounded by different amounts of impervious surface. Moreover, the gardens differ in vegetation and groundcover composition and structure, but because they are all community gardens, differences in composition and structure are assumed to be relatively similar between regions. The gardens range in size from 444 to 15,525 m2 in California and from 54 to 8778 m2 in Michigan, and are separated by approximately >2 km in California and >0.5 km in Michigan. All of the gardens have been cultivated for 1 to 47 years and do not use chemical pesticides and insecticides.

2.2. Ladybird Beetle Sampling

To assess ladybird beetle communities in the gardens, we used visual and trapping methods in both regions. In California, we sampled for adult beetles with visual surveys and sticky traps within 20 m2 plots at the center of each of the 18 gardens six times during summer 2014 (17–20 June, 7–10 July, 27–30 July, 19–21 August, 8–10 September, 29 September–1 October). Within the 20 m2, we visually surveyed vegetation and ground cover for adult beetles in eight randomly placed 0.5 × 0.5 m sub-plots. We collected all individuals observed and stored them in vials with ethyl alcohol. At four random locations within the plots, we also placed a 3″ × 5″ yellow sticky trap card (BioQuip Products Inc., Compton, CA, USA) on a galvanized wire stake for 24 h. In Michigan, we sampled for adult beetles by visually surveying five sentinel potted pea plants (Pisum sativum var. Dwarf grey) placed at each of the 13 gardens in Washtenaw County. Any ladybird beetle adults on plants were counted and identified to species. In addition, we swept surrounding vegetation in gardens for adult ladybird beetles using 10 full sweeps of a 0.2 m diameter net. All Michigan gardens were surveyed twice a week from 14 May to 20 July 2012. Sampling effort was consistent in all sites in each region throughout the sampling periods: in Michigan, the same two researchers conducted the sampling within the respective area for 30 min; in California, the same researcher conducted the sampling within the respective area for 60 min. The slight differences in sampling methods and years sampled between regions introduces some limitations discussed later in our conclusions.

We identified all ladybird beetles on traps and in vials to species using identification guides [39] and online resources [40,41]. Total abundance for each site for each species, total species richness, and total species diversity (Shannon's Diversity Index H) was tabulated across the months. Species diversity includes the relative distribution of species' abundances and was calculated using the vegan package in R [42]. For the analysis, we categorized species present in both California and Michigan as "shared species", and categorized species that were not both present in California and Michigan as "unique species".

2.3. Urban Landscape Analysis

To measure current levels of urbanization and to assess urbanization history, we summarized (1) the mean percent impervious surface surrounding gardens, and (2) the rate at which percent impervious surface has increased over time. For both regions (California, Michigan), we used the package "raster" in R (v 3.4.1) [43,44] to calculate the mean percent impervious surface within buffers of 10, 100, 500, 1000, 2000, 3000 m spatial scales surrounding each garden site based on land cover data from the US Geological Survey's National Land Cover Database (NLCD) 2011 Percent Developed Imperviousness dataset [45]. Here, a high total percent impervious surface indicates higher degrees of urbanization, and a low percent impervious surface indicates low degrees of urbanization. To calculate the rate of percent impervious surface change over time (henceforth "urbanization rate"), we collected this data at three time periods, as provided by the NLCD: 2001, 2006, 2011. Urbanization rate was calculated as the slope of the regression across these three time periods.

2.4. Statistical Analysis

We ran species accumulation curves to test whether species richness had been sufficiently sampled in both California and Michigan. The expected number of species in each geographic region was calculated using a sample-based rarefaction method known as the Mao Tau estimator [46]. Both regions showed evidence that richness was sufficiently sampled, exhibiting saturation in their species accumulation curves (Figure S1).

We first modeled abundance and richness for each region at multiple spatial scales to determine the best scale at which ladybird beetles respond to urbanization. We built seven generalized linear models (GLM) at 0, 10, 100, 500, 1000, 2000, 3000 m spatial scales assuming Poisson error distributions for count data. The model with the lowest Akaike Information Criterion (AIC) was selected as the best spatial scale for each region [47].

Urbanization rate was calculated by taking the slopes of linear regressions between time and impervious surface (NLCD: 2001, 2006, 2011) for each garden at a scale of 500 m. This was the buffer scale determined earlier to be significant for Michigan. California beetles best responded to impervious cover at 100 m, but at this scale urbanization rate did not vary by garden. Thus, we only analyzed effects of urbanization rate on ladybird beetle abundance, species richness and species diversity at 500 m for both regions. We also ran Pearson's r tests between values of urbanization rate and impervious surface at both 100 and 500 m to test for correlations between explanatory variables. Urbanization rate and impervious surface were not significantly correlated (Table S1).

To determine whether ladybird beetles significantly responded to percent impervious surface or urbanization rates, we constructed GLMs at the spatial scale appropriate for the region and predictor variable as described above. Abundance and species richness GLMs assumed Poisson error distributions, and diversity GLMs assumed Gaussian. All GLMs were then fit by Laplace approximation and goodness of fit determined by Wald Z tests [47]. This is what we refer to in the text as Poisson and linear regressions.

3. Results

We found 16 ladybird beetle species in California and eight species in Michigan over the sampling periods across the regions (Table 1). Species diversity index values were higher in California (ranging from 1.3 to 2.5 for all species) than in

Michigan (ranging from 0 to 1.5). Only four species were shared by California and Michigan, including: *C. septempunctata, C. Sanguinea, H. axyridis and O. v-nigrum*. Thus, 12 species in California and four species in Michigan were unique to that region.

Total ladybird beetle species abundance, richness and diversity (shared and unique species) were best explained by percent impervious surface at a 100 m spatial scale in California, but were best explained at a 500 m spatial scale in Michigan (Table S2). Total ladybird beetle species abundance and species diversity significantly increased with percent impervious surface in California, but significantly declined with percent impervious surface in Michigan (Table 2; Figure 2). Total species richness also generally increased in California and decreased in Michigan with percent impervious surface (Table 2). The divergent trend between regions was similar for shared species: shared species abundance, richness and diversity significantly increased in California with greater impervious surface, but were not significant for response measures in Michigan (Table 2). Unique species abundance also significantly increased with impervious cover in California, while unique species abundance significantly decreased in Michigan (Table 2; Figure 2). Of note, impervious surface cover gradients were comparable between California and Michigan.

In response to the rate of urbanization surrounding gardens, ladybird beetle abundance and species diversity increased with faster urbanization rates in California

(Table 2; Figure 3). Whereas, in Michigan, it was not significant for all response measures (Table 2; Figure 3). Species diversity of shared ladybird species significantly increased with faster urbanization rates for both regions (Table 2; Figure 3). The abundance of unique species significantly increased in California and decreased in Michigan with faster urbanization rates (Table 2; Figure 3).

4. Discussion

The influence of urbanization on biodiversity can change with environmental (regional) context. This comparative study between two urban regions in the US—California and Michigan—shows that organisms respond differently to urbanization depending on region. Ladybird beetles have a contrasting response to the intensity of urbanization as well as the rate at which urbanization occurs in different regional contexts, and we found only one unidirectional relationship between species diversity and urbanization rate between regions. The contrasting response is most apparent in the abundance of all species and unique species. We hypothesize that the effect of urbanization on unique species is driving this divergent pattern.

Our first question was whether the relationship between urbanization and ladybird beetles in urban agroecosystems changes with environmental context. We found that urban gardens are supporting more abundant and diverse ladybird beetle populations in more urban areas in California, while in Michigan, ladybird beetles in urban gardens decline in abundance, species richness and diversity with increasing amounts of impervious cover and faster urbanization rates in most instances.

Urbanization is clearly driving the abundance, species richness and behavior of ladybird beetles in California, as we have found in previous studies [29,50]. However, we show that this is not the case in another environmental context (Michigan). Though not specifically sampled in urban gardens, a majority of taxa decline in abundance and species richness with urbanization [2]. This is particularly apparent for vertebrates [18], but is also often the case for invertebrates [51,52,53]. In contrast, plant species generally increase with urbanization presumably because non-native species introductions outweigh extinctions in this group and because plants have smaller geographical ranges than mobile organisms with high dispersal abilities [54]. Organisms with larger ranges may be more sensitive to urbanization because urbanization can fragment migratory corridors and impede dispersal [55,56]. Given that we observed divergent geographical responses to urbanization most strongly for unique ladybird species and one similar response to urbanization by shared species, differences in dispersal ranges could possibly explain our results. If unique beetles to California have larger geographical distributions—often related to species dispersal ability and range size [57]—than beetles unique to Michigan, our results would be consistent with the dispersal hypothesis. However, we did not find strong evidence for this hypothesis in our results, because the reported geographic distribution for these species is relatively narrow for California beetles versus Michigan beetles (Table 1).

It is important to note, however, that though general trends in taxonomic responses to urbanization exist, all taxa that have been examined at multiple spatial scales or contexts exhibit some degree of divergence in responses to urbanization (6.9 to 33.3% of studies in a given taxon report different responses to urbanization depending on context) [2,28,30,58,59]. At larger spatial scales, urbanization is correlated with dense human populations that also coincide historically with nutrient-rich and biodiverse regions [60,61]. McKinney suggests that this can produce an apparent positive effect of urbanization on species abundance and richness [2]. Moreover, the longer periods of warm temperatures due to urbanization (i.e., urban heat island effects) may increase insect population abundance because of increased reproductive capacity [62], a common physiological response for arthropods [63]. At smaller spatial scales, local effects including management intensity and the destruction of habitat and pollution may impose negative effects of urbanization on species abundance and richness [64]. However, our results do not support this hypothesis given that ladybird beetles responded negatively to urbanization at larger spatial scales in Michigan, and positively at smaller spatial scales in California (Figure 2, Table 1).

The rate of urbanization, not only the amount of impervious surface, was important for explaining beetle abundance and diversity but exhibited different patterns depending on the group. Interestingly, while the abundance and diversity of unique species similarly diverged in regional responses to urbanization rate as to amount of impervious surface, shared species all showed positive unidirectional responses to urbanization rate. The predictor variables are not significantly correlated, and thus could theoretically have divergent effects (Table S1). Urbanization rates were actually relatively similar in California and in Michigan (at 500 m), which could explain why shared species had similar responses in each region while unique species had opposite responses. The differences in ladybird biodiversity between regions is therefore likely best explained by a species-level response: specifically, the response of unique species to California versus Michigan. Indeed, the strongest pattern from our study is the response of unique species to both the percent impervious surface and to the rate of urbanization, with abundance of unique beetles significantly increasing in California but decreasing in Michigan. This suggests that there are environmental filters at regional as well as local scales for species' traits that allow them to thrive in more urban areas and habitats [12,65] in California, and that in Michigan, those species are not present. Only species with traits that allow them to persist in urban environments should similarly increase with increasing rates of urbanization across regions. Indeed, traits including habitat, diet breadth and foraging efficiency have explained the global expansion of the ladybird species Harmonia axyridis into urban areas [66,67]. The similar unidirectional response of shared species to urbanization rate in our study supports this hypothesis.

Thus, our results may be explained by the legacy of land use change in each region and species' life histories/traits. California gardens have more species and more unique species that are not found in Michigan. Michigan gardens have fewer species, and 50% of those species were also found in California gardens. Most of the shared species across regions are aphidophagous (eat aphids), while in California the unique species to the region largely eat mites, scales and fungus (Table 1). For example, the fungus feeder *Psyllobora vigintimaculata* is very abundant in California and has a different life history than other species (Figure S2). The presence of species with these feeding preferences could be because of the industrial agricultural crops grown within and near our urban garden sites in California, historically and currently. Some of the region was once an orchard landscape, known as the "Valley of Heart's Delight", that has historically grown diverse fruit and nut trees [68]. Fruit trees and landscaped shrubs are often prone to scale, mite and mealybug pest damage along with crops like strawberries and tomatoes [69]. As discussed earlier, human populations preferentially settle in biodiverse areas (or "biodiversity hotspots") [61,70], and cities can coincide with threatened species distribution [71,72], possibly explaining positive relationships between ladybirds and urbanity. The legacy of agriculture in turn has permanent effects on ecosystems, and the influx of nutrients and irrigation can also create biodiversity hotspots [60]. Natural enemies were historically introduced from e.g., New Zealand and Australia for biological control of pests in the orchards. For example, the Dusky ladybird beetle*, Nephus binaevatus*, was released into California from New Zealand in 1922 to help control mealybugs in orchards [73]. This non-native species is unique to the California garden sites, particularly in very urban sites in Santa Clara county that were once orchards. Our sampled garden sites in California contain fruit trees such as citrus and stone fruit. Thus, some of the unique ladybird beetles like *N. binaevatus* and also *Coccinella septempuctata* are legacies of agricultural industrialization and urbanization unique to California, and/or may be present in gardens due to the availability of their prey/host.

The ladybird beetles in California—particularly those that are unique—may be more likely able to withstand environmental disturbance because they have been historically used in human-dominated systems for e.g., biological control. Non-native species to a region often have a greater ability to survive in a variety of habitats—including disturbed habitats—than native species [74]. In disturbed urban landscapes, more abundant species are habitat generalists and/or non-native to a region [2,75]. These species have ecological traits that allow them to exploit resources and persist [76], and environmental filters have been used to explain taxonomic differences between urban habitats [12,77]. Most (though not all) of the ladybird species that we observed in Michigan sites are native species from the greater Northeast and Midwest region (Table 1). This could explain why abundance and richness of beetles declines with urbanization in Michigan: many of the species are native and are less likely to survive and adapt to environmental disturbances like urbanization as non-native and invasive species [78].

Abiotic factors associated with urbanization and different environmental contexts may better explain the contrasts in ladybird diversity patterns in California and Michigan agroecosystems. Ladybirds must avidly consume water (e.g., dew, rain) for their survival [79], and water availability often drives their movement ecology and life cycle [80]. Thus, climate patterns (temperature, precipitation) can affect ladybird distribution [39,79], and significant climatic contrasts between regions could explain divergent patterns in ladybird abundance and species richness in gardens. In comparison to Michigan, which has a temperate climate with four defined seasons, California has on average warmer temperatures throughout the year and two seasons, one dry (summer; April to September) and one wet (winter; October to March). In California, urban gardens in the summer dry months (with <1 cm precipitation per month much in the form of fog) may provide an important irrigated habitat in comparison to the surrounding urban matrix. Indeed, supplemental irrigation in cities maintains and supports biodiversity during drought events [9,81,82], as California experienced during the time of this study. Michigan has an opposite precipitation pattern with relatively wet summers, receiving on average 8 to 9 cm/month in the summer [83]. Thus, limited water availability in the urban matrix may not affect arthropod distributions as strongly in Michigan. On the other hand, urban gardens in California may act as critical sources of water, food, and shelter and promote greater ladybird beetle foraging and fecundity rates [79], ultimately increasing their populations. If this is the case, then urban gardens in California may function as sources of populations rather than sinks, as has previously been suggested for biodiversity in urban habitats [1,25,84,85]. To test this hypothesis, we suggest examining habitats outside of gardens in these respective regions and their local conditions (in water availability, prey) to see whether ladybird abundance and diversity conservation is greater within gardens versus outside of gardens in natural habitats [86] and if it is explained by local conditions.

The slight differences in sampling methods and years sampled between regions are caveats in our study. Specific sampling methods as well as annual variation within regions may influence differences between regions. Moreover, although we only focused on landscape factors to better understand the ecology of urban agroecosystems across regions, local garden attributes including vegetation characteristics could also be important for species distribution [87,88]. Indeed, these caveats are generally the cruces of comparative studies and synthesis research. Although such studies are critical, comparing biodiversity assessments from multiple regions and research groups are inherently challenging to conduct and to analyze due to inconsistencies among methodological approaches in ecology, and research funding and timing. Given these limitations, this work shows strong relationships between landscape factors and local biodiversity with available comparable data across two regions. This is a first step towards a greater understanding of regional effects on urban biodiversity.

5. Conclusions

Ladybird beetles provide important pest control services and understanding their responses to urbanization can inform sustainable agricultural management in urban gardens. In this comparative study using urban agroecosystems as a field system, we show that organisms like ladybirds exhibit opposite responses to urbanization in different regional environmental contexts. Thus, we cannot expect universal responses of organism groups to urbanization. Based on previous research, region-specific responses to urbanization appear ubiquitous. Spatial scale is currently presumed to have a positive correlation with species abundance and diversity in increasingly urban environments. Here, we show that at least for ladybird beetles that is not the case. We suggest that region-specific responses are more likely due to the environmental factors (abiotic, biotic), urbanization history of the region, and the natural histories of species unique to each region. The natural history of species that are unique to urban agroecosystems could impart insight into how urban areas may be designed to conserve species that are more sensitive to urban environments. Future urban agroecology research should seek to draw more comparisons across distinct regions rather than generalizing responses of biodiversity to urbanization.

Supplementary Materials

The following are available online at http://www.mdpi.com/2071-1050/10/6/1829/s1, Figure S1: Species accumulation curves, Figure S2: Histograms of ladybird species sampled, Table S1: Correlations between explanatory variables, Table S2: Ladybird beetle sensitivity to percent impervious surface at various spatial scales.

Table 5-1. Ladybird beetle species sampled in California and in Michigan. We present: the respective region the species was found in, their feeding habits, the ecological role that they play in agroecosystems, their nativity in their respective region [39,48,49], and their current geographic distribution in the USA [39–41]. (CA = California; MI = Michigan; NA = North America).

Species	**Region Observed**	**Feeds on**	**Ecological Function in Agroecosystems**	**Origin**	**Distribution in US**
Adalia bipunctata	CA	aphids and mites	predator/pest control	native	West coast, Northeast, few Midwest records (historically most of US and Canada)
Coccinella californica	CA	mostly aphids	predator/pest control	native	West coast CA
Cycloneda polita	CA	mostly aphids	predator/pest control	native	West coast US and British Columbia
Hippodamia convergens	CA	mostly aphids	predator/pest control	native	Throughout US and western Canada
Hyperaspis quadrioculata	CA	aphids and scale insects	predator/pest control	native	Central to south CA
Nephus binaevatus	CA	aphids and scale insects	predator/pest control	non-native	Central to south CA
Psyllobora vigintimaculata	CA	fungus	fungus and mildew control	native	Throughout US and Canada
Scymnus cervicalis	CA	mites and scale insects	predator/pest and mite control	native	East US to south CA
Scymnus coniferarum	CA	mites and scale insects	predator/pest and mite control	native	CA and scattered west NA records
Scymnus marginicollis	CA	mites and scale insects	predator/pest and mite control	native	CA to British Columbia; scattered NA records
Scymnus nebulosus	CA	mites and scale insects	predator/pest and mite control	native	South CA to Canada

Stethorus punctum	CA	mites and scale insects	predator/pest and mite control	native	West coast US; Northeast, west to north Great Plains
Coleomegilla maculata	MI	mostly aphids	predator/pest control	native	East NA to southwest US
Cryptolaemus montrouzieri	MI	mites and scale insects	predator/pest and mite control	non-native	Throughout US
Hippodamia variegata	MI	mostly aphids	predator/pest control	native	Northeastern to middle US and Canada
Propylea quatuordecimpunctata	MI	mostly aphids	predator/pest control	non-native	Throughout NA (native to the Palaearctic)
Coccinella septempunctata	MI, CA	mostly aphids	predator/pest control	non-native	Throughout NA (native to the Old World)
Cycloneda sanguinea	MI, CA	mostly aphids	predator/pest control	native	West to south CA; NC and FL
Harmonia axyridis	MI, CA	mostly aphids	predator/pest control	non-native	Throughout US and southern Canada, except northern Rockies
Olla v-nigrum	MI, CA	mostly aphids	predator/pest control	native	Throughout US, except ME and Pacific Northwest

Table 5-2. Results of regressions predicting ladybird beetle abundance (AB), richness (RI) and Shannon's Diversity Index (SH) as a function of percent impervious surface (IS) and urbanization rate (UR). Scale indicates the spatial scale in meters used to calculate predictor variables IS and UR. Coefficients and P values are derived from Wald Z tests, which assess goodness of fit of generalized linear models to data assuming Poisson error distributions (AB, RI) or Gaussian error distribution (SH).

Dataset	Region	Scale	Predicted	Predictor	Coefficient	*p*-Value
All	MI	500	AB	IS	−0.015	0.01
All	CA	100	AB	IS	0.019	<0.001
All	MI	500	RI	IS	−0.018	0.08
All	CA	100	RI	IS	0.009	0.06
All	MI	500	SH	IS	−0.013	0.02
All	CA	100	SH	IS	0.006	0.05
Shared	MI	500	AB	IS	−0.001	0.92
Shared	CA	100	AB	IS	0.031	<0.001
Shared	MI	500	RI	IS	−0.016	0.29
Shared	CA	100	RI	IS	0.021	0.05
Shared	MI	500	SH	IS	−0.003	0.54
Shared	CA	100	SH	IS	0.007	0.09
Unique	MI	500	AB	IS	−0.023	0.004
Unique	CA	100	AB	IS	0.018	<0.001
Unique	MI	500	RI	IS	−0.020	0.16
Unique	CA	100	RI	IS	0.004	0.46
Unique	MI	500	SH	IS	−0.005	0.44
Unique	CA	100	SH	IS	0.003	0.76
All	MI	500	AB	UR	−3.524	0.15
All	CA	500	AB	UR	2.231	<0.001
All	MI	500	RI	UR	0.932	0.77
All	CA	500	RI	UR	2.292	0.11
All	MI	500	SH	UR	1.372	0.55
All	CA	500	SH	UR	2.331	0.04
Shared	MI	500	AB	UR	3.710	0.22
Shared	CA	500	AB	UR	1.925	0.29
Shared	MI	500	RI	UR	4.665	0.23
Shared	CA	500	RI	UR	2.024	0.50
Shared	MI	500	SH	UR	2.698	0.09
Shared	CA	500	SH	UR	3.110	0.02
Unique	MI	500	AB	UR	−10.88	0.01
Unique	CA	500	AB	UR	3.376	<0.001
Unique	MI	500	RI	UR	−4.020	0.47

Unique	CA	500	RI	UR	1.705	0.33
Unique	MI	500	SH	UR	−1.506	0.48
Unique	CA	500	SH	UR	1.473	0.24

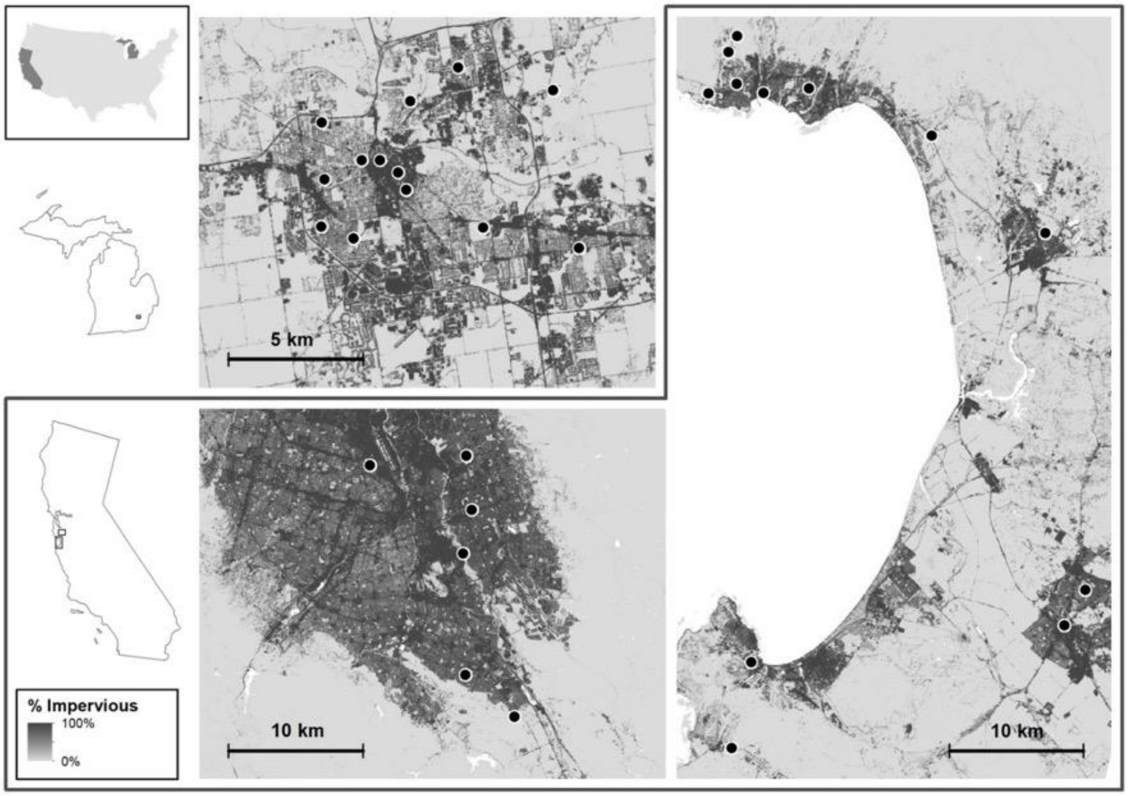

Figure 5-1. Study regions in Michigan (a) and in California (b) where community gardens (black circles) were sampled. Increasing percent impervious surface (NLCD 2011) shown with increasing shaded color.

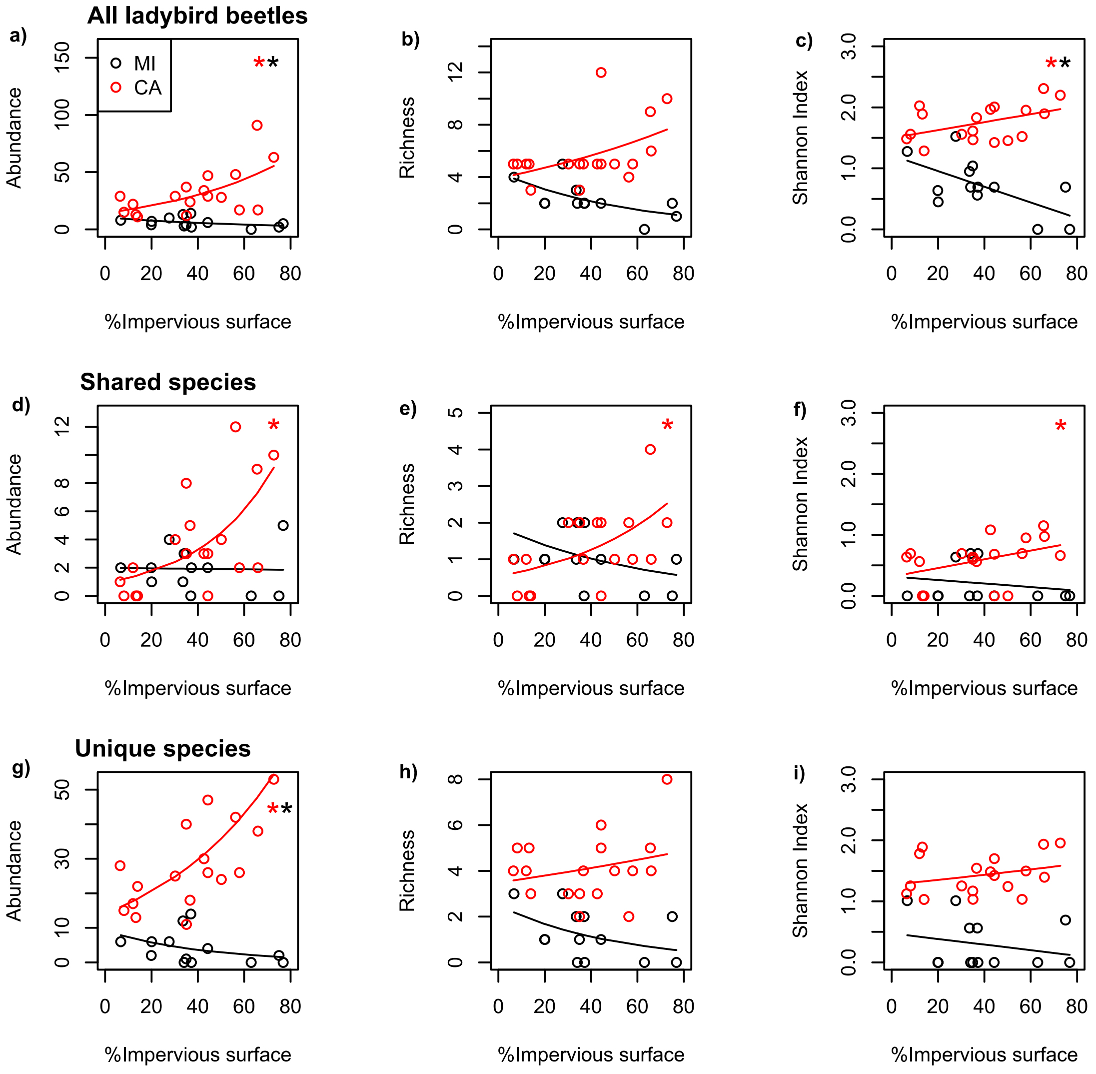

Figure 5-2. Effect of impervious surface on abundance, richness and diversity of ladybird beetles. Regressions of abundance, richness and diversity (Shannon Index) of Michigan (black lines and points) and California ladybird beetles (red lines and points) as a function of % impervious surface at 500 m for MI and 100 m for CA. All species combined (a–c, top row), species that are shared by both CA and MI (d–f, middle row) and species unique to each region (g–i, bottom row). * indicate

significant regressions ($p < 0.05$). In (b), Poisson regressions for MI and CA are partially significant ($p < 0.10$).

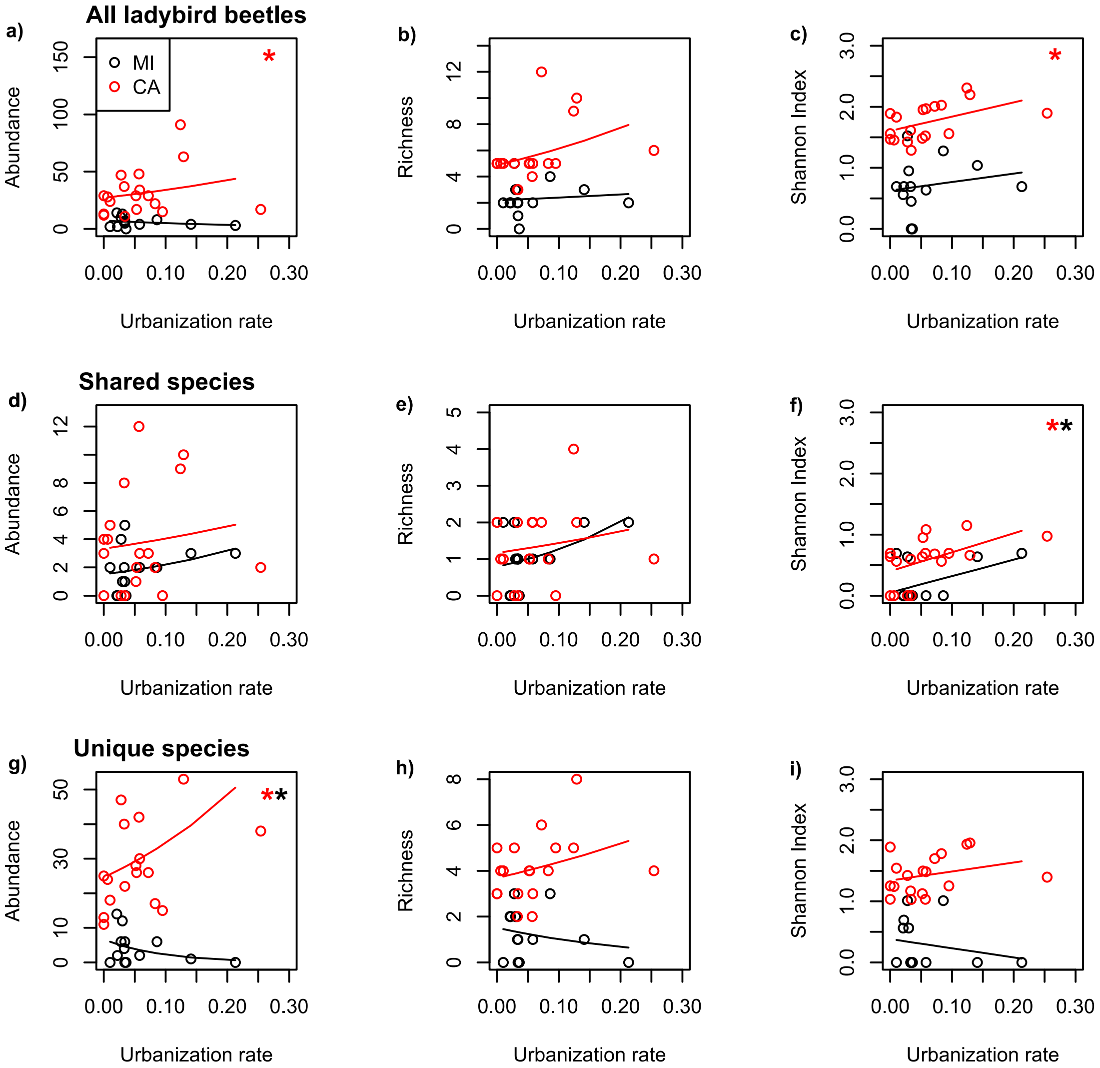

Figure 5-3. Effect of urbanization rate on abundance, richness and diversity of ladybird beetles. Regressions of abundance, richness and diversity (Shannon Index) of Michigan (black lines and points) and California ladybird beetles (red lines and points) as a function of urbanization rate at 500 m. All species combined (a–c, top row), species that are shared by both California and Michigan (d–f, middle row) and species unique to each region (g–i, bottom row). * indicate significant regressions ($p < 0.05$).

Part II

6. Water use behavior, learning and adaptation to future change in urban gardens

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Published in *Frontiers in Sustainable Food Systems*, 2018

Abstract

Urban agriculture is undergoing a contemporary global renaissance, providing fresh food for growing urban populations and vital environmental benefits for cities. Despite urban agriculture's social-environmental importance, a rural bias in agricultural research has left critical gaps in our understanding of how urban agroecosystem management can sustainably produce food in the future. Specifically, there is a need to study urban agriculture water management due to recent drought events, likely increases in urban water scarcity, and higher temperatures. Gardeners can play a decisive role in increasing urban agriculture's sustainability through their water, soil and vegetation management. Here, we examined water use, vegetation, and soil management in the California Central Coast - a region facing drought - to better understand how urban agriculture management affects water use sustainability. We worked with gardeners to study their water management decisions using citizen science, where volunteer gardeners collected their own water use behavior data and participated in a survey to describe their behaviors around water use, water conservation, and plot level management. We found that water use varies by gardener, and water use is positively related to mulching and crop cover in plots.

Contrary to expectations, gardeners that reported high levels of concern over current environmental conditions (drought) and water conservation in the survey tended to use high amounts of water, suggesting that environmental worldviews do not necessarily translate into everyday practices. On the other hand, gardeners in gardens with more rules and regulations around water use tended to use less water, highlighting the practicality of enforcing rules and regulations during drought periods. Gardeners reported interest in adapting gardening practices to more sustainably use water regardless of their current practices. The combination of education and rules and regulations will help improve the sustainability of garden systems in times of environmental change.

Keywords: urban agriculture; citizen science; water management; drought; California

Introduction

More than half of the world's population now lives in cities (UN, 2015), and urban agriculture is a resource to enhance urban food system sustainability (Wiskerke, 2015). Urban agriculture is broadly described as the production of crop and livestock on small parcels of land within cities (Ackerman et al., 2014; Zezza and Tasciotti, 2010), and encompasses home/private gardens, production-focused farms, community/allotment gardens, roof-top gardens, and community orchards (Mougeot, 2000). Urban agriculture may be an essential social resource for food security and nutrition in food desert neighborhoods (Alaimo et al., 2008; Walker, 2007), but also an ecological resource in cities experiencing loss of natural habitat and related environmental issues including biodiversity loss and storm water run-off (Ackerman et al., 2014; Gittleman et al., 2017; Lovell, 2010; Mandel, 2013). Urban agroecosystems are thus critical social-ecological systems for urban food systems and ecosystem services broadly (Barthel et al., 2015; Wiskerke, 2015). Despite urban agriculture's growing relevance as agroecosystems in cities and as components of urban food systems, a rural focus in agricultural management has left a gap in our understanding of how urban agroecosystem resource management influences sustainable food production in urban systems (Lin et al., 2015). Urban gardeners may lack appropriate agroecological knowledge of practices to promote sustainable resource management of plants, soils and water (Gregory et al., 2015). This is important because plants in agroecosystems are carefully maintained through soil nutrient management and water supplementation based on plant ecophysiology and environmental conditions (Daryanto et al., 2017; Prasad et al., 2008), but current and future patterns of extreme drought and heat in cities are increasing the need for natural resource inputs while simultaneously reducing access to them (Hunt et al., 2013; Milly et al., 2008). This could limit the sustainability of urban agriculture due to negative effects of drought on crop production (Tardieu et al., 2000) and for natural resource conservation in cities (Eriksen-Hamel and Danso, 2010).

Water access and availability often challenge urban gardeners because both environmental limitations and city policy mechanisms regulate the continuous access to water in gardens and therefore crop productivity (Gregory et al., 2015). In California, a recent five year-long drought spurred cities to implement severe watering restrictions for urban community gardens (Community Gardens Program, 2016), limiting the number of days and hours that gardeners could access and use water. Gardeners may not have experience or the resources to adapt their growing practices to changes in temperatures and water availability to reduce their water consumption while still supporting plant productivity (Oberholtzer et al., 2014). Plant sensitivity to the negative effects of drought-induced high temperatures on soil moisture retention (Blum, 1996; Monneveux et al., 2011; Prasad et al., 2008) may prompt gardeners to reevaluate soil and water management. Some gardeners may mulch beds to retain soil moisture and change watering methods in attempts to reduce water use and maintain plant survivorship (Gregory et al., 2015), while few change the types of plants that they grow (Egerer et al. 2019). Some gardens may implement rules and regulations in attempts to reduce water use (Turner, 2011), but it is unclear if rules reduce overall water use. There is a critical need for research in urban agriculture on limited resources that links scientists with practitioners to improve learning and actionable science (Lin and Egerer, 2017; Ossola et al., 2017). Urban gardeners can play a valuable role in urban agriculture research and in the production of knowledge that gardeners can utilize to use water more sustainably – reducing water use and increasing efficiency through behavioral change. Involving gardeners in the design and undertaking of research can be an approach that promotes sustainable water management through gardener learning and behavior adaptation to overall improve food system sustainability. This can bolster garden resilience – the ability of the social-ecological system to respond to environmental and social disturbances (Carpenter and Folke, 2006) – in a changing, increasingly more extreme climate.

Learning and knowledge exchange is key to building resilience and garden sustainability (Schultz et al., 2015; Westley et al., 2013). Adaptation of management practices through experience and learned behavior can affect the short and long term resilience of gardens to resource scarcity and environmental change (Barthel and Isendahl, 2013). Experimentation, behavioral adaptation and learning in agroecosystem management prepares gardeners for current and future disturbances and therefore their ability to handle change (Krasny and Tidball 2009, Barthel et al. 2010, Barthel et al. 2015). Participation in resource management can empower gardeners to make management changes through their learning and interacting with natural resources and garden social networks (Okvat and Zautra, 2011). Including gardeners in scientific research on agroecosystem resource management has great potential to co-produce knowledge on sustainable management (Gregory, 2018) in systems experiencing environmental change (Childers et al., 2015). Research approaches like citizen science that are shaped by civic ecology frameworks can better incorporate ecology, stakeholder experience, and policy for "action-based" research to build community resilience (Krasny et al., 2014; Shirk et al., 2012). Citizen science entails public participation in organized research project data collection (Louv et al., 2012); the knowledge generated from the research can be collaboratively produced by citizen scientists and researchers, informed by diverse experiences and perspectives, and directly applied by practitioners involved in knowledge production (Grove et al., 2016; Shirk et al., 2012). Thus using citizen science under a civic ecology framework integrates the realms of science, engagement and education (Krasny et al., 2014; Shirk et al., 2012), and sits at the nexus between scientific discovery and social change (Krasny and Tidball, 2015).

In this research, we draw from civic ecology frameworks and citizen science approaches in order to understand drivers of water use in urban gardens to improve the sustainability and resiliency of urban agricultural systems under environmental change. In addition to the social learning benefits described above, citizen science lends itself to urban agriculture research because of the practical challenges of the research (e.g., site access, difficulty of measuring inputs and outputs) (Pollard et al., 2017). We worked in a drought-stricken region – the California Central Coast – and asked three questions: 1) What are the reported biophysical variables and social variables that influence gardener water use behavior in urban gardens? 2) Do plot-level biophysical variables and social variables predict self-reported water use by gardeners? 3) What do gardeners learn and how do they report changing their water, soil, and vegetation management to be more sustainable through research participation? To answer these questions we used a contributory citizen science model with collaboratory model features (*sensu* Shirk et al. 2012): scientists designed the research questions and asked for volunteer gardeners to contribute data, but scientists and gardeners collaboratively collected and interpreted data on management. Both groups reviewed the results, provided iterative feedback, and together brainstormed solutions to water access and availability challenges facing gardens. With this research, we aimed to uncover relative unknowns in urban agricultural research: water usage and drivers of water use behavior.

Hypotheses

Our study had three hypotheses and we had a priori expectations informed by past research. H1) Water use varies by biophysical factors that influence gardeners; here we expected that gardeners will water more in hotter temperatures and when having more crops. H2) Water use varies by social factors; here we expected that gardeners that convey more concern about environmental changes occurring in the region (drought, water scarcity) will be more conservative in their water use as will gardeners in gardens with more watering rules. H3) Gardeners will learn about water use through participation; here we expected that gardeners would report knowledge of water use and types of changes to their practices.

Methods

Study system

We worked in four urban community gardens in the Central California Coast, one in Monterey Co. (36.2400° N, 121.3100° W), two in Santa Clara Co. (37.3600° N, 121.9700° W), and one in Santa Cruz Co. (37.0300° N, 122.0100° W). The cities range in size, with estimated population densities of 50, 274, and 232 people/sq. km, respectively (2010–2014 U.S. census period) (US Census Bureau, 2014). The current populations of major cities included in the survey range from 1.03 million (San Jose) to 22,000 (Marina). The study took place from August to September 2017 (Aug 1 – Sept 15, 2017), a time of year typically characterized by little to no rainfall, periodic heat waves, and drought conditions (Rippey, 2017). In some counties there were water restrictions, and garden bylaws for some gardens had influenced or required the garden management to impose watering restrictions, limiting the number of days in the week and time of day that gardeners were allowed to water (Community Gardens Program, 2016). It is within these gardens that water is particularly a resource concern for the management. In each of the four gardens, we monitored five individually managed plots, for a total of 20 plots across all gardens. Plots had similar irrigation infrastructure; participants had either a personal or shared spigot with hose attachment at or near to their plot. In each plot, we collected data on vegetation characteristics, ground cover characteristics, soil properties, ambient temperature, rainfall, and on water used by gardeners. For each gardener, we collected data on their perceptions of water use and learning. We collected a portion of the data (vegetation, ground cover, temperature, etc.) and then worked with volunteer citizen scientists in each garden to collect other data (water use, rainfall). We describe both of these data collection processes in detail below.

Citizen science water use data collection

We used a citizen science approach with volunteer gardeners (henceforth "participants") as it is an appropriate and effective research tool to collect urban gardener data (Pollard et al., 2017). The gardens in which we worked were interested in understanding their water usage and wanted to make efforts to reduce water. For this reason, participants wanted to work collaboratively with us to inform the design of the data collection by providing feedback on data collection feasibility and by providing feedback and self-assessment throughout the study period. The collaborative aspects of the research aimed to better provide participants the opportunity to produce knowledge that influences their agroecological practices and resource management (Sharp et al., 2011).

For our study, we solicited five participants per garden to participate in the study using paper fliers hung in the garden and through communication with management. Participants had to be active gardeners (i.e., growing plants, regularly visiting the garden) and be willing to collect data for the project's entire duration. Participants either contacted the researchers if they were interested in participating or the researchers asked the gardeners if they were interested in participating during site visits. We included three interested gardeners that had participated in our pilot study the previous year (Lin et al., 2018) in the present study. Our sample size was limited by logistical feasibility and efforts to maintain high quality communication with all participants across the counties, and introduces a limitation to our study.

Each participant was given a Gardena water meter to measure their water use (Gardena Brand Electronic Garden Hose Water Meters), and we demonstrated as well as provided instructions on how to read these meters. The meters were installed directly to each of their faucets at the plot level so that they only recorded their own water use. We provided participants a clipboard and a data sheet to record the amount of water they used each time they watered, at what time of day, precipitation at their plots and the weather conditions at the time of the watering. We encouraged other notes on their watering experience. At the end of the six weeks, the participant gave data sheets to the research team. One simple rainfall gauge was installed onsite to confirm that there was no rainfall during the experimental period. Participants checked on this when they were taking their measurements.

We reviewed data quality for the participant-collected data before data analysis. We eliminated from the analysis two individuals who ceased watering because of crop death from extreme heat events (we qualitatively discuss their survey responses below). We calculated three water use variables for each participant for the analysis: 1) water use per watering event (i.e., each time a participant watered) per meter square of gardening surface area (L/m^2); 2) mean water use across all watering events per meter square area (L/m^2); and 3) coefficient of variation in water use across all events (CV). Thus in this study, water use is the amount of water withdrawn by gardeners (i.e. not crop consumptive use). Water use variables were standardized for area because garden plots were of slightly differing sizes, and rescaled because water use was quite variable across participants and therefore rescaling was needed for optimal model convergence.

Garden plot characteristics data collection

Vegetation and ground cover characteristics

We measured garden plot characteristics on vegetation cover, ground cover, soil properties, and temperature at each of the 20 plots to assess the types of management used at the plot level. We did this to determine whether there were certain biophysical characteristics of the plots that would be predictive of water usage aside from what participants self-reported (described below). For plot vegetation, we measured the number of crops in the plots and the percent cover of crops, weeds, and herbaceous plants in the plots. For ground cover, we similarly measured the percent cover of mulch, straw, rocks, grass and bare soil in the plots. For both vegetation and ground cover, we used a visual assessment and an estimate of the percentage of total plot surface of each type of vegetation and ground cover covering the plot. In addition, we reviewed the crops growing in the participants plots and classified the crop species by their water use needs using the US Farmers Almanac watering guidelines (https://www.almanac.com/), and the University of California Agricultural Extension Service (http://cagardenweb.ucanr.edu/). We then created general classifications for the entire crop community as needing low, medium or high water users based on these “ideal” guidelines (similar methods to Pataki et al. 2013) in relation to the percent crop cover measured in participants’ plots. We use these classifications to determine whether crop water needs predicted water use.

Soil properties

We collected baseline data on soils in the plots, including two bulk density cores for each plot as well as five soil cores up to 15-20 cm in depth to analyze for soil organic matter, water holding capacity, bulk density, and texture. This was done in part to provide participants with information about their soils. We followed Wilke’s (2005) standardized methods for soil property measurements that are used for assessing amended soils in urban gardens (e.g. Grewal et al. 2011). To determine soil organic matter (SOM), we used the Loss on Ignition (LOI) method (500ºC, 4 h) with dried soils to calculate the percent SOM. To determine bulk density, we weighed fresh soil, dried samples at 105ºC, for 24 h, and then calculated bulk density with the final dried weight. To determine soil water holding capacity (WHC), we followed the standardized method that determines the maximum amount of water retained by the soil against gravity by saturating soil samples, draining soils of free water, and evaluating only the water held by the soil (Wilke, 2005). We filled 2 x 2” cylinders with a perforated base with sieved, fresh soil, and placed them in a water bath overnight. We then capped and placed cylinders on a tray of sand for approximately 6 h, allowing soils to drain, and then removed and dried soils (105ºC, 24 h) to calculate WHC. For soil texture, the proportion of sand, silt and clay were determined through a particle size analysis by A&L Western Agricultural Laboratories (Modesto, CA, USA).

For the analysis (Section 2.5.1.), we selected only one soil variable due to significant correlation among soil properties. We chose the soil organic matter content (%) because it not only had the best fit for the model and was correlated to other soil properties (e.g., WHC), but many participants reported that they use mulch and compost in their plots which is likely reflected in the soil organic matter content in this system (Egerer et al., 2017).

Climate measurements

We monitored the ambient temperature of each participant’s plot with a temperature logger (Onset HOBO UA-001-08) placed 1.3 m above the plot to record temperatures (°C) directly at the plot. The loggers took temperature readings every hour over the study period. We checked the temperature loggers throughout the study period to ensure that they were in good working order. We downloaded and collected the data at the end of the study period, and quality checked and cleaned the data. For each plot, the temperature at the time of the reported watering event was taken for each event to examine if ambient plot temperatures predicted water use.

Survey of participants for self-assessment of water use and learning

We took a four step approach to understand participant's perceptions of their water use behavior. The first step was to understand whether participants understood their water use before the project, and we distributed paper survey questionnaires to participants at the beginning and end of the study period. In the pre-study survey we asked them how much water they think that they use each time they water, including an option of "I don't know". This question was important because it provided baseline information on participant knowledge. The pre-study survey also asked participants what crops they are growing, their watering frequency, what variables influence their water use (e.g., rules, weather, plants), what they add to their soils (e.g., compost), and their perceptions of current drought and climatic conditions. We also asked the three participants that participated in our pilot study the previous year (Lin et al., 2018) whether participation had changed their watering since. The second step was to monitor the amounts of water that participants use based on the self-reported water data for each participant. The third step was to ask participants again in a post-study survey how much water they think that they use after their research participation and data collection. The post-study survey asked participants how much water they think that they use each time (given their data collection), if and how useful they found participating in the project (1-5 scale and open response), what learning outcomes they gained from the project, what they discussed/shared with others through their participation, and the influence of participation on their water management decisions and behaviors. In addition, the post-study survey asked participants to describe if and how they changed their water management in the research process. We used this survey data to compare how much participants know about and whether they gained a better understanding of their water use.

The fourth step was to send participants their results in the form of a tabulated water use summary and personalized soil profile six weeks after the study's end. In this report we provided summary statistics across all participants so that they could gauge how their use fits within the range of data collected. To collect qualitative data and to provide an opportunity for participants to suggest how to improve the study, we included a survey form with the summary results soliciting further thoughts on their experience and on their results after six weeks. We also asked participants what could be improved from the research/ data collection design. We used this questionnaire to collect any additional information not captured by the pre- and post-study surveys, to gain more insight into post-study impact, and for participants to inform future research design. We calculated summary statistics for the pre- and post-survey responses, and reviewed all qualitative responses to inform our analysis and interpretation.

Data analysis

Self-reported watering data and stated influences on watering behavior

We used the collected survey responses to help inform which explanatory variables would go into water use models. Information from the survey suggested that: 1) self-reported watering behavior of participants may be influenced by biophysical variables such as the local weather and by the plants in plots; 2) self-reported watering behavior may be influenced by a social variable: garden rules and regulations on watering; 3) participants are heavily amending the soils and ground cover in their plots; and 4) participants are divided on views on environmental conditions with those voicing high concern to those voicing little to no concern for drought and water scarcity in the region. We therefore used four quantitative biophysical non-correlated explanatory variables informed by the survey to model water use behavior across four categories: one climate/weather variable (plot-level ambient temperature (°C) at the time of the reported watering event), one crop management variable (percent crop plant cover per 1 m^2), one soil management variable (percent organic matter), and one ground cover management variable (percent mulch and straw cover). We also selected the presence of garden rules and the level of environmental concern reported as social variables for the model. For the rules variable, we classified participants as: 1) having rules in their garden, where the garden limits the number of days and times gardeners can water, or 2) having no restrictions on water use. For the environmental concern variable, we coded participant pre-study survey responses by either conveying high concern for regional drought and water scarcity, or having little or no concern.

Mean water use and water use CV

To determine whether variables strongly predict water use, we modeled the three water use response variables in our analysis at the scale of the participant (mean water use (L/m^2), water use CV, and water use per event (L/m^2)). For the mean water use and water use CV analysis, we built generalized linear regression models (GLMs) assuming log-normal error structure with the four biophysical variables and two social variables (2.5.1) and their potential interaction because rules and environmental concern may interact to influence watering outcomes. We used GLMs because it is a flexible multivariate approach (Zuur et al., 2009) that could test what biophysical variables and social variables best predict self-reported water use. We used the *glmulti* package and function (Calcagno and Mazancourt, 2010) in the R statistical environment (R Development Core Team, 2016) to determine best fit models. Models with the lowest Akaike Information Criterion (AICc) values were selected as the best fit models and model assumptions were checked with Shapiro-Wilks tests (Bolker et al., 2009).

Water use per event

For the water use per event analysis, we used generalized linear mixed models (GLMM) with a link log function and Gaussian error distribution and repeated measures to model the liters of water used per event (L/m^2). Participant nested within garden site were treated as random effects, and the biophysical and social explanatory variables (2.5.1) were treated as fixed effects. This approach allowed us to: fully examine the distribution of the data; test the importance of each and combinations of variables for predicting amount of water used at an event; and include participant and site as random effects (Zuur et al., 2009). We rescaled the response variable due to the wide range in water use amounts per event (e.g. 1 L – 1000 L). We used model comparison with AICc to evaluate the best model fit, considering the best model with the lowest AICc score (Burnham and Anderson, 2002). GLMM analyses were performed using the *lme4* package (Bates et al., 2015) in R.

Water use in relation to crop needs

To examine whether crop water needs predicted water use, we used GLM to compare mean water use among participants for crop water need groups. We fit the model and ran a post-hoc test using the *glht* function in the *multcomp* package (Hothorn et al., 2008) to assess difference in water use. Mean water use per event (L/m^2) was the response variable, and low, medium and high crop water needs were the predictor variables.

Self-assessment of water use and gardener learning

To evaluate whether gardener understanding of their water use matched gardener watering management practices, we reviewed the pre- and post-study survey responses in relation to the water use data that participants collected. We compared pre- and post-study survey responses to the survey question asked in both surveys, "How much water do you think that you use each time?", to evaluate differences in water use behavior understanding and thus potential learning outcomes. In both surveys, participants could provide an estimated numerical value (in gallons or liters) or could select "I don't know" as a response. Based on the reported responses we categorized participants as those that provided a numerical response, and those that replied, "I don't know" or did not reply. Of those that reported a response, we categorized participants that (1) approximated their water use and it was close to their actual measured mean water use (± 20 liters, 10% of the average water used over the study period) as having a better understanding, (2) approximated and overestimated their use (> 20 liters), (3) approximated and underestimated their water use (< 20 liters).

We qualitatively analyzed the participant responses to the open-ended question of "What have you learned from participating in this study?" on the post-study survey to examine if and how participants had a better understanding of their watering practices. We reviewed participant responses to open ended questions on what they learned, how they found the study useful, and their experiences and thoughts on project participation. We summarized scaled (1-5) values for the question "How useful did you find participating in this project?" Moreover, we reviewed participant responses on how to improve their water conservation, and put it in the context of our own experience as researchers working with participants through the course of the study.

Ethics statement

All participants gave their informed verbal and written consent for inclusion before they participated in the study. The study was conducted in accordance with the University of California-Santa Cruz (UCSC) Office of Research Compliance Administration. The research was exempted from Institutional Review Board under #HS2569. The UCSC operates under a Federalwide Assurance approved by the DHHS Office for Human Research Protections, FWA00002797.

Results

Water use behavior and garden plot characteristics

Self-reported watering data

Self-reported water use behavior varied by participant with participants using on average 202 L or 31 L/m^2 for each watering event (i.e., time that they watered) over the course of the six weeks (Table 1). A majority of participants watered their plots 1-2 times per week (53%). The other participants that watered their plots more frequently (daily or 3-4 times per week) tended to use on average less water per event.

Stated influences on watering behavior

In the pre-surveys, garden-set rules, regulations, and limits were the most frequently cited influences on participant watering behavior (65%). This was followed by the weather (40%), plant needs (35%), other gardeners (15%), and the news/media (e.g., reports on drought and climate) (10%). Survey responses also mentioned that participants manage their soils and ground cover by adding compost, straw and mulches to their plots, and how this may influence their watering because they assess the soil (e.g., "I eyeball the soil [dryness] to know how much water to use"). Participants documented little to no rainfall, just dew and fog at their plots, and this was confirmed by regional climatic data during this period that recorded just 2 mm of precipitation in Santa Cruz, California and 1 mm of precipitation recorded in San Jose, California for the entire study period (US Climate Data, 2017).

Predictors of mean water use and water use CV

The best model predicting mean water use over the study period included mulch/straw ground cover, soil organic matter, garden rules and environmental concern variables (Table 2). Mean water use was significantly higher with greater mulch/straw cover in plots (Table 2; Figure 1). The best model predicting water use CV included garden rules, environmental concern and their interaction, with water use CV significantly increasing among participants in gardens with rules and with low environmental concern (Table 2; Figure 2).

Predictors of water use per event

The best models predicting water use per event included the percent mulch/straw cover, temperature at watering, and environmental concern (Table 3), followed by models also including garden rules and soil organic matter. Participants that expressed high concern for environmental changes in the region like drought, heat and water scarcity tended to use more water per event than those that expressed little to no concern, and participants in gardens with more mulch ground cover and more crop cover tended to use more water (Figures 1, 2). Participants tended to use more water when watering at higher ambient temperatures (Figure 1). At this scale, the interaction between environmental concern and garden rules was relatively insignificant, and did not improve model fit (Table 3).

Water use in relation to crop needs

Gardeners had crops across the range of water use needs including crops that have low water use needs (deep-rooted crops; e.g., asparagus, tomato, melon) as well as crops with high water use needs (shallow-rooted crops; e.g., lettuce, corn, cabbage) (Supplementary Table 1). Water use did not vary depending on recommended watering needs of plants in the post-hoc comparison.

Self-assessment of water use and gardener learning

Self-assessment of water use

Gardener expectations of their water use often differed from their measured water use. In the pre-study survey, 70% of participants reported that they did not know how much water they use each time. In the post-study survey after project participation, 65% of participants estimated an amount of water that they use each time. Yet 35% did not know or did not respond indicating that they did not learn about their water use through participation. Of the 65% that estimated, more than half (69%) of the participants had a better understanding of their water use, estimating close (± 20 liters) to their mean water use while 8% of participants overestimated and 23% underestimated mean water use. These quantitative values were further supported by qualitative survey responses in the post-survey. One participant clearly articulated the general difficulty of assessing their water usage even after 6 weeks of monitoring: "I [still] do not have a concept of what the amount (the actual number) of water I am using looks like."

Gardener self-reported learning

Nevertheless, a majority of gardeners found participation in the study to be useful, rating a 4 (47%) or 5 (26%) out of 5 for the post-study survey question "How useful did you find participating in this project?" In the analysis of the post-survey response to the question "What have you learned from participating in this study?" participants reported that they learned several valuable lessons through their participation. We identified two main learning processes from the participant responses: 1) monitoring water use is a learning process through which participants better understand how to identify how much water is used (more technical/ mechanical); and 2) monitoring water use is a learning process through which participants learn about themselves and the people around them (observational).

Gardeners that learned about the technical facts of water use included those that simply stated "I learned it takes a lot of water to grow food" or "I learned about water usage." Another participant reported: "I learned that the pressure of the water reflects how much water is used and how long I have to water." Other participants in this group associated their learning about water use specifically with the study methods, design and equipment. Stated by one participant: "I learned I need to watch the length of time it takes to water, the water flow rate, and how frequently I am watering. Keeping a record helped a lot with this." Another participant described how "the materials used, the equipment to track your water - how much going out - is really interesting. I use 10 times more than I thought I was." This participant also reflected learning about themselves, and about other gardener's practices through observation, and reported that "[the study] told me that I can look at others' plots and see what they are doing (and what they are not)." Here, the participant related learning about sustainable water use in community gardens to direct observations of others, and they then linked this back to self-reflections on their behavior. Another participant that indicated learning about themselves stated: "I change my watering a lot daily and weekly depending on weather and seasonal plantings."

Some participants reported a "ripple effect" both within the garden and an effect beyond the garden. Stated by one participant: "I like participating in a group effort. Everyone participating has learned something from it. There's a ripple effect." Another participant highlighted that: "This project helps me serve as an example to other community gardeners. The collected water measurements strengthen my case for water conservation [in the garden]." Expanding impacts beyond the garden, one participant said that participation "has and will change my habits at home, too." Participants brought forth deeper realizations on their experience and on broader implications with passing time in the weeks after the study. For example, one participant voiced that "the garden can use less water than what it is using. We could further limit the number of days [that we are allowed to water]."

Gardener management adjustments

Participants reported that they are changing their management to improve the characteristics of their soil and reduce their water use. One participant actually implemented a drip irrigation system towards the end of the study in order to better control her use, illustrating a change in water use behavior. Several other participants also voiced a plan to implement drip irrigation in their plots. Participants reported plans to improve their soils after receiving their summary results. Stated by one participant: "I am making a plan to improve my soil profile and texture by using more compost and also reduce watering…I would like to amend my soil and hopefully be able to have it retested to see if my soil profile and soil texture improve." However, this was not universally voiced by participants, and a few participants – particularly those in gardens with rules and regulations – stated that they did not change their watering practices since the study's completion. Other participants did relay that the soil results provided new information for the whole garden: "I was surprised to see the sand, silt and clay proportions in the soil. At the garden, we generally talk of our soil as clay but we actually have more sand than clay and are unaware of silt content."

Gardener study feedback

Last, participants expressed their opinions on how to improve the study's design, data presentation, and tangible management outcomes of the study. Participants conveyed that they still require more information to understand the results, specifically in the management of "healthy" soils. Participants wished for specific recommendations regarding soil properties that they believe are necessary for healthy soil improvement. For example, one participant stated: "I would like to know more about what is included in [soil] "organic matter" and how this relates to fertilizers and compost. I am also aware that there needs to be a healthy microbe population in the soil but not much more than that. Mostly I'm interested in how I can tell the soil is healthy – what to look for in texture, smell, water retention, ease of planting, etc." Other participants reported interests in learning about best methods of watering without erosion and more information about the specific implications of certain soil properties (texture, nutrients) for water management.

Discussion

We worked with urban gardeners in a region (California) experiencing environmental change to better understand current urban agriculture water use, and to generate knowledge needed for urban agriculture's future sustainability. In our research, we found that citizen science gardeners use water conservatively and generously over the season depending on the agroecological characteristics in their garden plots and the rules in place at their gardens, but not necessarily with their conveyed environmental concerns. Gardener narratives revealed that water and garden soils are generally poorly understood, and that gardeners misperceive the amount of water that they think that they use. Although most participants water based on what they think their plants need, the lack of relationships between plant watering needs and water used may point to gaps in information on water use, and that gardeners may be using more or less water than is necessary for growing the plants. Water seems to be an intangible component of urban agroecosystems that is challenging to quantify and to conserve by individuals. Garden implemented rules could reduce water use, and gardeners can improve their understanding of their water use by participating in data collection. Many of the participants could estimate their water usage post-study, and most reported research participation useful and conveyed specific learning outcomes. Many of these participants are also eager to adapt their management – particularly of their soils – to be more sustainable.

We explore two key patterns that we see in our analysis of participant water use in these gardens: 1) gardeners generally misperceive their water use and can use high amounts of water in their plot systems regardless of plant needs. But these generous water users also use water-conserving practices (mulching with woodchips and straw) and report strong concerns about environmental changes occurring around them (extreme heat and drought); and 2) gardeners with garden-implemented rules on water usage tended to use less water, suggesting that governance systems could regulate water use if people will not reduce water use willingly. Reports from the participants suggest that education and learning through, for example, voluntary water monitoring can also be an effective tool towards water conservation in absence of rules, although it may take more time and effort.

Pattern 1: Environmental concerns do not necessarily lead to environmental behavior

Participants in the study used over double or triple the recommended amount of water for gardening in the region (~25 L/m^2/week) (UCCE Master Gardener Program, 2014). Plant needs influenced 35% of participant watering, even though many of the participants that cultivated plants with low water needs used greater amounts of water. Although we did not ask about plant selection or measure indicators of plant performance, a lack of a pattern among crop types and water use suggest that gardeners could be using more water than necessary for crop survival and production. Counter to expectations, low water use was not evident in environmentally concerned participants. Participants that expressed concerns about the climate and environmental changes like drought and extreme heat did not overall use less water, rather their use was variable and many used high amounts of water per event.

A contradiction in worldview and watering behavior supports theories in environmental psychology. Theories of planned behavior (Ajzen, 1991) and Values-Belief-Norm theory (Armitage and Conner, 2001; Harland et al., 1999; Steg and Vlek, 2009; Stern and Dietz, 1994) support that worldview beliefs do not necessarily predict consistent behaviors because other normative beliefs and attitudes also affect behaviors (Ives and Kendal, 2014). Behaviors are multiple and changeable, thus different value "orientations" (e.g., nature centered vs self-centered) that shape people's beliefs may produce similar environmental behaviors and vise versa (i.e., similar value orientations may produce quite different behaviors) (Ives and Kendal, 2014). Studies on household water consumption show that attitudes expressed towards water conservation are not representative of water consumption (Aitken et al., 1994). In domestic gardens, residents' practices, attitudes and beliefs often contradict because gardens are leisure spaces, rather than spaces where environmentally sensitive practices are enforced (Askew and McGuirk, 2004). Everyday practices like water use reveal inconsistencies between beliefs and practice due to conflicts among worldview beliefs and urban place-making processes that are associated with habitual behaviors (Allon and Sofoulis, 2006). Water use may be better explained by habits related to differences in the perceived necessity of caring for plants than by water-saving beliefs conveyed by people.

Participants using high amounts of water in our study are actually using more sustainable practices including adding straw cover and compost that would likely warrant less water use. Research in urban agriculture soils encourages composting, cover cropping and straw mulching to improve soil fertility and water holding capacity (Beniston and Lal, 2012; Gregory et al., 2015) because ground cover and soil management methods can reduce the rate of soil moisture loss (de Pascale et al., 2011). Increasing the application of water-saving ground cover and soil amendments should suggest reduced water use. Yet we found that participants are adding a lot of inputs to their plots across a number of management factors and may be misunderstanding the synergies among inputs. These participants seem to use more inputs – using more water, more straw, and more compost – despite their crop types. Such results also support the above assumption that many gardeners manage for plant care without fully understanding the interactions among their sustainable management decisions.

Considering whether to focus on the practice or the process through which management decisions are made, the results suggest that while gardeners may be eager to focus changes on particular practices, gardeners may need to focus on the process through which they make water use decisions to make impactful changes. Conveyed environmentally concerned worldviews and intentions to use less water may not translate to behaviors and are therefore not a good proxy for water conservation outcomes.

Pattern 2: Rules can reduce water use

The participants that had rules at their gardens as to what days and hours they could water used relatively less water. This suggests that rules and regulations on water usage can reduce gardener water use by reducing the frequency of intensive watering or by instilling a notion of shared norms around water where one are expected to use less by the greater social community (Seligman and Finegan, 1990). Community expectations and governance systems in place to conserve water may reduce water use through "good citizenship" notions (Holmes, 1999). The effect of rules may be explained by indirect effects through relationships among watering practices, plant needs and temperature. Gathered from their water use logs, the participants in the gardens with rules were watering at cooler times of the day – in the morning and evening as required by the management – and are therefore potentially subconsciously using less water because of the cooler ambient temperatures. By influencing when gardeners water, rules may have a more indirect effect on water use through the effect of cooler temperatures on water use behavior. In addition, these participants happened to be growing fewer water-intensive plants in their plots (i.e., tomatoes rather than lettuce), although this was not required by the management and probably due to chance. These participants may perceive lower water use requirements and water less. These participants also had less crop cover in their plots, and we observed several of them targeting water usage at the specific plants rather than the overhead spraying that we observed by participants with highly vegetated plots.

We caution against universalizing watering rules and regulations for all gardens, however, because rules may have short-term impacts but not produce long-term effects on water conservation, and because water governance is complex itself.

First, the participants in gardens with rules did not report adjusting their practices after project participation. Second, there are nevertheless problems with high water use at these gardens: the garden managers in gardens with specific garden-level rules disclosed information that the rules do not universally reduce water use by all gardeners, and that there are instances when other gardeners (those not participating in the study) use high amounts of water on watering days to compensate for reduced water access. This occurs especially for plots with high water need crops such as corn. Water governance can elicit grievances among garden members and management (Egerer and Fairbairn, 2018; Turner, 2011) and should be carefully considered and implemented. Managers and garden organizations should design water use rules and regulations based on the watering needs of the plants in the garden, but importantly also on the social context of their gardens and knowledge of water management held by their gardener communities because there are differences in the cultures and conceptualizations of water and plant needs (Allon and Sofoulis, 2006; Head and Muir, 2007; Jackson, 2006; Turner, 2011).

Gardener knowledge and adaptation potential in complex social-ecological systems

How can urban gardeners adapt their agroecological practices to improve the sustainability of resource use in changing climates and during times of resource stress (e.g., water shortage, drought)? In this study in four community gardens, we show gardeners variably use water across environmental concerns and crop regimes, and that these gardeners differ in their perception of their water use. The relatively small sample size limits the predictive power of our statistical models, and the study design limits exploring other interaction effects because we did not select gardeners based on their backgrounds or beliefs, rather on their interest to participate. Yet we show that water usage is individualized based on the gardener's perceptions of climate, needs of plants, and the water governance structures of their gardens. Our findings further suggest that gardeners may not fully understand their water use and the water requirements of the plants that they are growing. Water is difficult to conceive for gardeners as indicated by the disparity in actual water used, the perception of use and the recommended needs of the plants. This is similar to findings in domestic gardens and landscapes that argue that water is a misunderstood natural resource (Kolokytha et al., 2002) that supports the pleasures of everyday domestic life but is in tension with environmental worldviews on conservation (Head and Muir, 2007).

Monitoring water usage through participation in research can build the capacity of urban gardeners to learn about their water use through multiple learning processes and adapt their management practices with changing environments. We suggest that research engagement and monitoring may reveal to participants the inconsistencies in their water use behavior in relation to their worldviews, and may provoke better understandings of water use that may generate long-lasting changes that improve garden resource use sustainability. Management can incorporate opportunities for learning about water use as well as water use conservation techniques and strategies for gardeners into their governance plans. Education could be an effective way to have gardeners willingly reduce water use over time, with more knowledge sharing among gardeners as a way to build long-term sustainable water use networks in gardens, rather than rules and regulations that may only inhibit water use short term during drought events.

Gardeners are eager to learn and adapt their gardening through their learned experience in tandem with more “formal” assessments and recommendations provided by researchers (Gregory et al., 2015). In our study we found that while some participants’ learning processes were more technical or straightforward (e.g., “I learned I use more water than I expected”), others learned about themselves and about how their behaviors relate to others’ through observation. Participating in the research project had an overall positive impact on the garden community and catalyzed conversations on sustainable water use and conservation. Participants expressed that they were excited to be a part of a team effort to reduce water use, or that they have suggestions for the entire garden after their participation. As evidenced in participant narratives after the study, participants are using the soil analysis paired with their water usage numbers provided by the researchers to make management changes in their plots (“I am making a plan to improve my soil profile and texture by using more compost and also reduce watering”).

Gardeners also want to know how other gardeners performed and if and why they used water differently to give them a better understanding of their own management. Stated by one participant: “I would have liked to have seen some of the other gardeners results to compare if the purchased soil originally added to the boxes is holding fertility.” And another participant reported: “I was hoping to see more of an analysis of water use by [each] participant…It would be informative to see if our overall water use is average, or, differs from the other gardens studied (and why). Is our water holding capacity lower, average or high compared to other gardens? Is the percentage of organic matter and soil type roughly the same or vastly different? These would be useful to know in managing the garden." Thus educational outreach activities that will be of interest to gardeners can focus on soil properties in relation to management suggestions, and this is similar to findings by Gregory et al (2015) in New York City gardens. In addition, there was no relationship exhibited between plant water needs and water use. This highlights missing knowledge of plant needs, and outreach activities should convey information to gardeners about crop needs and crop/variety selection to reduce water use.

To reflect on managing the study and working with the participants, we experienced and observed both the challenges and the rewards of citizen science research. Maintaining consistent and clear communication with participants on project expectations, materials and methods (i.e., working the water use meter), and interpreting results was time consuming, as was cleaning and proofing the data for potential errors once it was collected. Nevertheless, our conversations with participants both during and after the study illuminated that gardeners are genuinely inquisitive, are experimental in their agroecological practices, and are motivated to learn about how to better their gardening for water conservation aims – even if their behaviors do not indicate such. As researchers, participants taught us about resourceful water conservation methods, and about how to creatively share results with practitioners. Research that engages both researchers and gardeners together can improve knowledge exchange between groups and facilitate actionable science (Ossola et al., 2017). Collaborations based on scientific integration and knowledge co-production can be a platform for research towards global sustainability (Mauser et al., 2013).

Conclusion

In this study, we investigated the biophysical and social variables that affect water use in urban gardens, what gardeners learn from water research participation, and how gardeners adapt their management to be more sustainable. Our results suggest that water use behavior requires a nuanced understanding by managers and researchers because gardens are socioecological systems in which interactions between biophysical variables, governance systems, and human behavior together shape water use. The inconsistency between human belief and behaviors is part of the complexity of working in social-ecological systems, and this complexity argues for the necessity to work in citizen science platforms where there is co-learning among researchers and resource managers. Working with gardeners in research expands an understanding of urban agriculture water use that can have on-the-ground positive effects on resource management and urban welfare through gardener learning with critical implications for the sustainability and resiliency of urban food systems.

Tables and Figures

Table 6-1. Descriptive statistics for the plot-level variables measured in the garden plots by gardener citizen scientists (A) and by researchers (B-D) over the duration of the study.

	Min	**Median**	**Max**	**Mean**	**SD**
A. Water use variables					
Mean water use (L)	35.4	212.6	2167.9	292.6	462.7
Mean water use per area (L/m^2)	1.3	10.4	104.5	31.4	35.3
Water use CV	0.05	0.5	2.2	0.6	0.4
B. Climate variables					
Plot temperature at watering event (◦C)	15.3	26.3	33.5	25.3	4.8
Mean plot temperature (◦C)	19.5	22.2	26.4	22.3	2.3
C. Soil variables					
% Organic matter	9.4	19.5	28.8	19.2	6.2
% Water holding capacity	16.8	28.8	50.7	30.1	9.7
% Sand	46.0	62.0	86.0	63.6	11.9
% Silt	8.0	24.0	36.0	22.4	8.2
% Clay	6.0	16.0	20.0	14.0	4.3
D. Vegetation variables					
No. of crop species	2.0	4.5	14.0	5.2	2.9
% Crop cover	7.0	22.1	85.0	27.8	19.9
% Bare soil cover	5.0	52.5	85.0	47.5	27.4
% Mulch/straw cover	0.0	12.0	88.0	26.4	41.1

Table 6-2. Generalized linear models (GLMs) that best predict amount of mean water used (A) and variation (CV) in water use (B) over the six-week study period (△AICc to null model: A) 16.5; B) 23.0). Parameters indicate the biophysical variables (measured in the garden plots), and social variables (determined from gardener surveys), with reference level in parentheses. Significance of variable as a predictor indicated in bold (significance assessed at $P \leq 0.05$). An "x" indicates an interaction.

Response variable	**Predictor variable**	**Coefficient**	**SE**	**t**	**P**	**AICc**
A. Mean water use (L/m^2)	Intercept	-0.63	2.88	-0.22	0.83	163.7
	Garden rules (yes)	-0.21	0.35	-0.61	0.55	
	Env. concern (low)	0.42	0.68	0.61	0.55	
	Soil organic matter (%)	0.73	0.53	1.37	0.20	
	Mulch/straw cover (%)	0.02	0.01	3.82	**0.002**	
B. Water use CV	Intercept	-0.88	0.21	-4.17	0.001	3.6
	Garden rules (yes)	0.22	0.26	0.85	0.41	
	Env. concern (low)	0.05	0.29	0.18	0.86	
	Garden rules (yes) x Env. Concern (low)	1.41	0.34	4.17	**0.001**	

Table 6-3. Generalized linear mixed models (GLMM) predicting water used by participants per watering event over the six-week study period, with participant nested within garden as a random effect. We present the top ten best possible models ranked by decreasing AICc. Models were composed of all possible combinations of biophysical variables (measured in the garden plots) and social variables (determined from gardener surveys) with reference level in parentheses. "Temp" is the plot temperature at the watering event (◦C); soil organic matter, mulch/straw cover, and crop cover are percentages measured within participants' plots. An "x" indicates interaction. To note, the interaction effect between social variables did not improve model fit at this scale.

Response	Model	AICc	△AICc
Water use per event (L/m^2)			
	Mulch/straw + Temp + Env. concern (low)	2101.65	0
	Mulch/straw + Temp + Garden rules (yes) + Env. concern (low)	2101.89	0.24
	Mulch/straw + Temp + Soil organic matter + Env. concern (low)	2102.38	0.73
	Crop cover + Temp + Env. concern (low)	2104.34	2.69
	Crop cover + Temp + Env. concern (low) + Garden rules (yes)	2105.69	4.04
	Env. concern (low)	2110.26	8.61
	Garden rules (yes) x Env. concern (low)	2112.40	10.75
	Garden rules (yes) + Env. concern (low)	2112.05	10.40
	Mulch/straw + Temp	2206.66	105.01
	Mulch/straw	2207.57	105.92

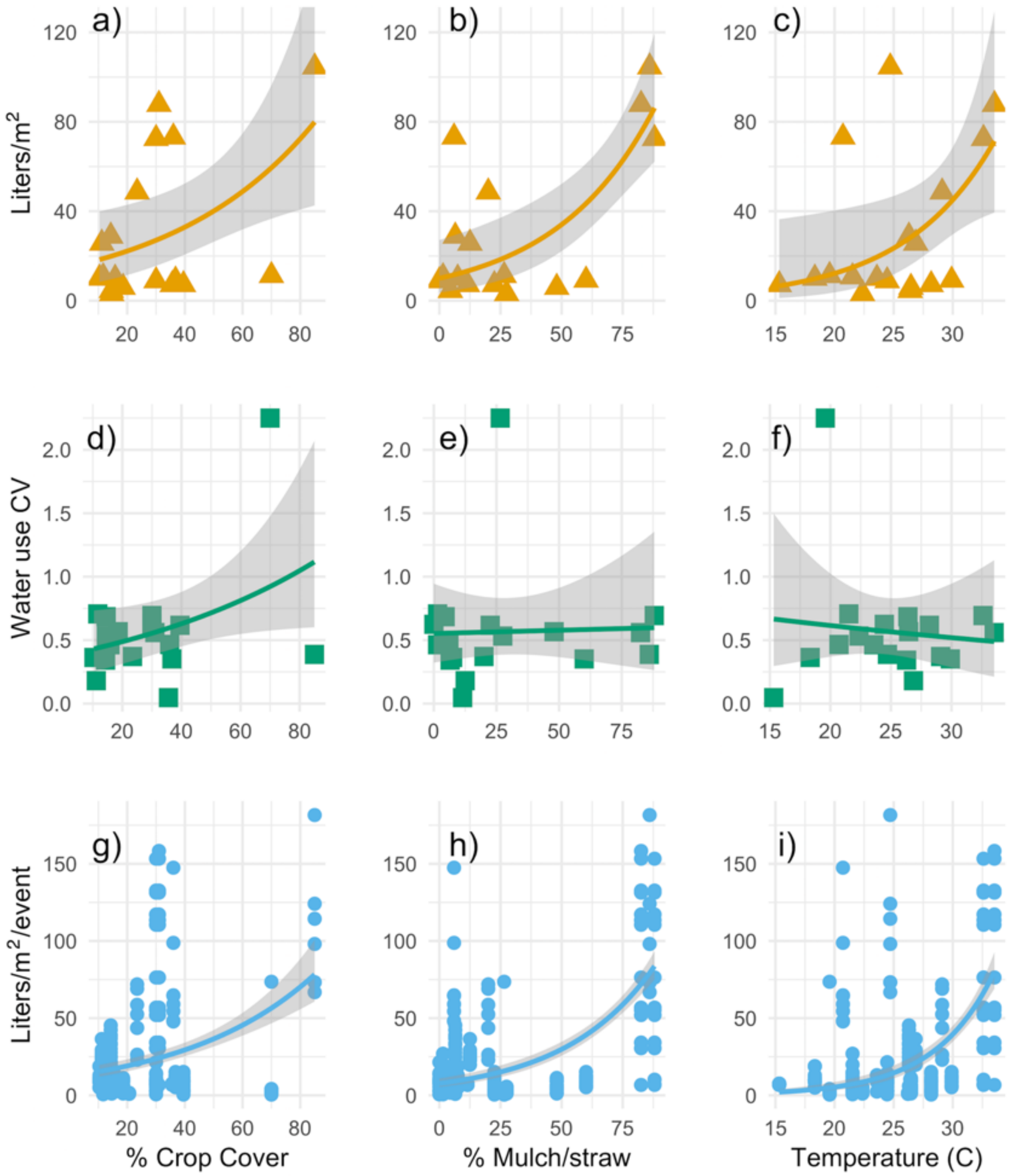

Figure 6-1. Relationship between garden biophysical variables (measured in the garden plots) and the three water use variables examined over the six-week study period: mean water use (a-c), water use CV (d-f), and water used per watering event (g-i). Gray shading and fitted line show model fits in Table 2 and 3 (i.e., GLM or GLMM).

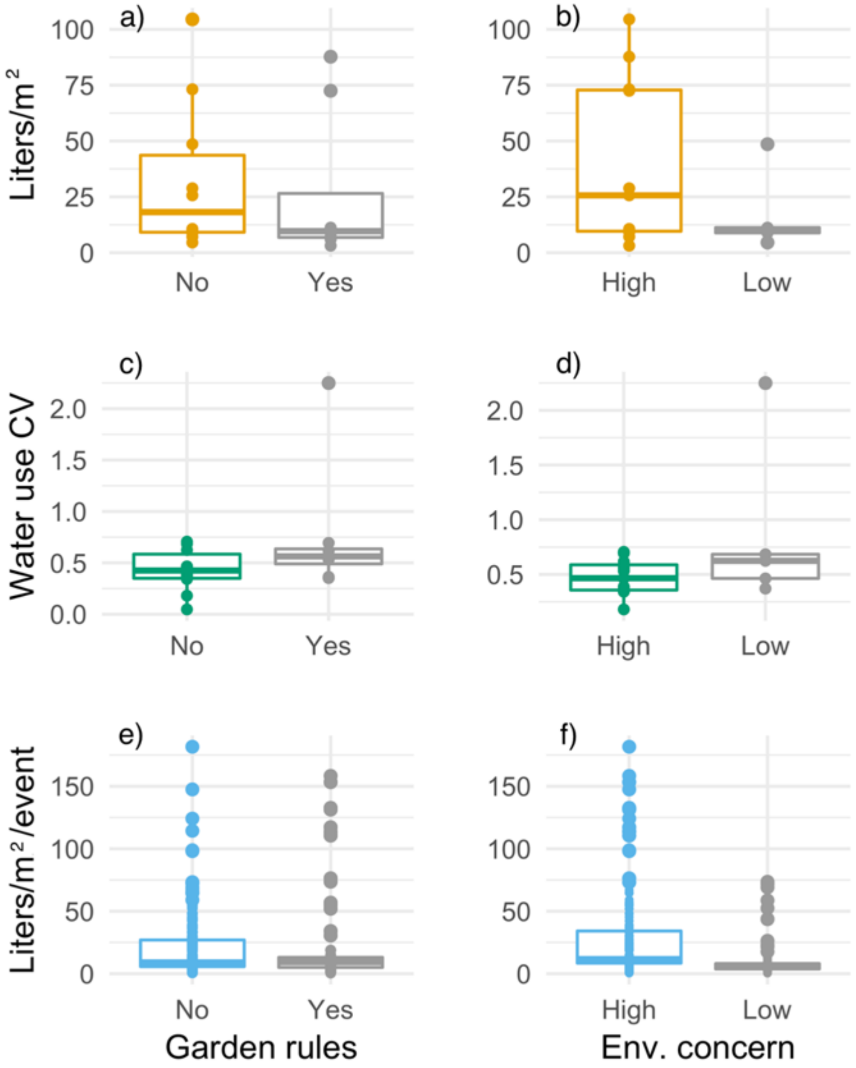

Figure 6-2. Relationships between social variables (garden rules; participant's environmental concern) and water use variables examined over the six-week study period: mean water use (a-b), water use CV (c-d), and water used per watering event (e-f).

7. Temperature variability influences urban garden plant richness and gardener water use behavior, but not planting decisions

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Published in Science of the Total Environment, 2019

Abstract

Urban environments are being subject to increasing temperatures due to the combined effects of global climate change and urban heat. These increased temperatures, coupled with human planting preferences and green space management practices, influence how urban plants grow and survive. Urban community gardens are an increasingly popular land use, and a green space type that is influenced by unique climate-human behavior interactions. Despite ongoing rapid temperature changes in cities, it is unknown how gardeners are adapting to these changes, and to what extent changes influence planting decisions and patterns of urban plant diversity. In this study, we monitored the variation in daily air temperatures and measured plant species richness at the garden and garden plot scale in 11 community gardens in Melbourne, Australia. We surveyed >180 gardeners to better understand the relationships between temperature variation, garden plant species diversity, and gardener management practices. We found that garden scale temperature variability is driven by regional context, and temperatures are more stable in landscapes with higher impervious surface cover. Gardeners agreed that climatic/temperature changes are influencing their watering behavior, but not their plant selection. Instead plant selection is being driven by desired food production. Yet, when comparing two bioregions, temperature did have a measurable relationship with garden plant composition in the region with more temperature variation. Temperature variability negatively related to plant species richness within garden plots, providing evidence that plant survival is related to climate at this scale in such regions. Although gardeners may be able to water more in response to regional climate changes, gardeners are unlikely to be able to completely control the effects of temperature on plant survival in more variable conditions. This suggests the inner city with more stable temperatures (albeit potentially hotter for longer due to heat island) may accommodate more species diverse gardens.

Keywords: climate change; temperature variability; urban greening; water use; urban agriculture; plant diversity

1. Introduction

Climatic gradients often predict species distribution across natural landscapes (Soberon, 2007). Species have temperature and moisture thresholds that allow or inhibit their survival in an ecosystem. Plants in particular are often found along temperature and moisture/precipitation gradients. Consequently, the distribution of plant species are changing with global climate change as temperatures become hotter, and in some places drought events become more extreme (Kelly and Goulden, 2008; Lenoir and Svenning, 2014; Neilson et al., 2005). This can limit plant water availability and thus survival (Breshears et al., 2005; Galiano et al., 2011; Martínez-Vilalta and Piñol, 2002; McDowell et al., 2010). Temperature is a strong predictor of species diversity in natural communities (Grubb, 1977) because of species traits related to species performance (Kleidon and Mooney, 2000; McGill et al., 2006). Climate extremes are having profound impacts on trophic interactions, food webs and the general ecology of regions (Brose et al., 2016; Tylianakis et al., 2008; Walther, 2010; Walther et al., 2002).

Human dominated environments such as cities, are often perceived to be shaped by drivers other than the climatic and biophysical drivers that shape natural landscapes. Human preferences influence resource management decisions that affect plant species distribution beyond natural bioclimatic barriers (Kendal et al., 2018). Vegetation within urban ecosystems is shaped by habitat transformation, as well as unique socioecological filters including biophysical conditions of the urban environment and individual human preference (Pataki et al., 2013; Williams et al., 2008). While, temperature gradients remain a strong filter of urban cultivated plant richness (Kendal et al., 2012a), supplemental irrigation and nutrients can be common in urban residential landscapes (Faeth et al., 2005) and allow some plants preferred by people to thrive through human intervention (Clarke and Jenerette, 2015; Hope et al., 2003; Jenerette et al., 2016). Within urban ecosystems, the diversity and distribution of plant species are therefore influenced by both environmental filters at a regional scale and local scale as well as through socioecological interactions at the level of the individual (Aronson et al., 2016; Avolio et al., 2015; Williams et al., 2008).

Changes in temperature and precipitation due to global climate change (Freitag et al., 2018) and intensifying urban heat island effects (Oke, 1973) are therefore likely to affect the composition and diversity of urban gardens (Eriksen-Hamel and Danso, 2010). Irrespective of human intervention in the form of irrigation and fertilizer application, higher temperatures and evapotranspiration are likely to affect the plant species grown in urban environments where they are sensitive to heat and water stress (Albrecht and Haider, 2013; Jenerette et al., 2016). In addition, more intense heat and drought may therefore affect the way that people use resources to manage urban green spaces such as gardens (Balling et al., 2008; Jenerette et al., 2013).

Urban gardens are places where there are unique and complex interactions between temperature, precipitation, watering behavior and plant selection. Urban gardening is a popular past time around the world (Galluzzi et al., 2010; Lawson, 2005; Mougeot, 2000; Zezza and Tasciotti, 2010), and is one of the important ways in which people interact with urban nature (Andersson et al., 2007; Egerer et al., 2018; Okvat and Zautra, 2011) and shape the plants of the urban environment (Galluzzi et al., 2010; Loram et al., 2008; Smith et al., 2006). Ambient temperatures in gardens can be influenced by region-scale urbanization as well as by local garden-scale plant cover, and this can also influence watering behavior (Lin et al., 2018). Greater amount of impervious surface cover surrounding and within urban gardens increases mean and maximum temperatures (Lin et al., 2018), probably because impervious surfaces retain heat due to low albedo (Oke, 1973). In contrast, greater plant ground cover and higher tree density is associated with cooler temperatures and climate mitigation within urban green spaces (Bowler et al., 2010; Gill et al., 2007; Huang et al., 2008; Shashua-bar et al., 2009) including within urban gardens (Piacentini et al., 2014). Local temperatures likely affect the degree to which plants are stressed in this managed environment (Eriksen-Hamel and Danso, 2010), due to effects of temperature on soil moisture retention (Craul, 1992; Pickett et al., 2011). Climate conditions and the potential temperature effects on plants within garden plots may lead gardeners to think that they need to supplement more or less water in response (Avolio et al., 2015; Lin et al., 2018). Yet we know less about how urban temperatures may affect plants cultivated in gardens and their care, as provided by gardeners, within and between gardens. If and how gardener resource management of water and plants within gardens responds to climate variability is critical to urban sustainability.

It is important to assess how climate variability – in the form of temperature fluctuations, extreme heat and drought conditions – may affect the composition and distribution of urban plant communities as cities, and therefore urban plant distributions, expand (Jenerette et al., 2015). It is of particular importance to understand these relationships in urban agroecosystems because variability in temperature, precipitation and their interaction significantly influence crop plant yield (Ray et al., 2015) and consequently ecosystem service provisioning. Increasing temperatures and drought patterns in urban environments will likely negatively affect crop plant productivity and survivorship in urban agriculture (Lobell et al., 2011) because of higher urban temperatures (Eriksen-Hamel and Danso, 2010; Kalnay and Ming, 2003) and water use restrictions on outdoor irrigation implemented during times of drought (Kendal et al., 2012b). If urban gardeners are unable to maintain crop irrigation during heat events, water limitation when plants are most susceptible to evapotranspiration can increase plant vulnerability to sun scorch, disease and pest damage (Gourdji et al., 2013; Meineke et al., 2013). Thus temperature and precipitation variability are still likely to affect species survival and distribution within urban garden plant communities, but there is still much to understand in the context of current urban environmental change.

In this study, we explore the relationships between temperature variability, urban gardener decision making, and plant species richness in garden plots in community gardens across the city of Melbourne, Australia. Community gardens, or gardens managed by a collective of individuals who are each allocated a plot, are popular in urban Melbourne, which is a city known for its temperature and precipitation fluctuations. Climate events over the past decade (e.g., the "Millennium Drought") indicate that climate patterns are becoming more extreme, in tandem with urbanization (Coutts et al., 2007). However, there is little knowledge of if and how this variability is experienced by green space managers, such as urban gardeners. There is also little knowledge of if and how gardeners are adapting to these proposed changes within their individual garden plots. We aim to fill this research gap through a mixed-method study that uses field-collected measurements of garden temperatures and garden plants at both the garden scale and at the individual plot scale, and quantitative and qualitative survey responses on gardener decision making at an individual level. We focused our study at these multiple scales because individual people do different things at their plots in the same garden. Thus, the plot scale and individual level analyses focus on individual gardener behavior; and the garden scale analyses focus on the response of temperature variability and garden plants to the local and regional context. Specifically, in this study we asked: 1) Do landscape (regional) and local (garden) factors predict urban garden temperature variability at the garden scale? 2) Can temperature variability in turn explain observed plant species richness in urban gardens at the garden scale or at the plot scale? 3) Does climate variability (temperature, precipitation) influence gardeners' reported planting decisions and water use behavior at an individual level? If not, what factors are important? This study fills an important gap in the understanding of relationships among temperature variability, plant species richness, and gardener behavior across a climatically variable urban environment. This is especially important as climate becomes more variable across the world, and in increasingly popular urban agroecosystems.

2. Methods

2.1. Study system

We worked in 11 community gardens distributed from east to west across the Greater Melbourne Metropolitan area in Victoria, Australia (study area center point: 37°50'8.60"S 145° 2'15.31"E) (Figure 1). Melbourne is the capital of Victoria, covers 9992.5 km^2 and has approximately 4.7 million residents (City of Melbourne, 2018). Greater Melbourne has the largest and fastest growing population in Australia (2.7% growth from 2016-2017) (Australian Bureau of Statistics, 2018). Melbourne’s climate is temperate and is generally considered highly variable (Bureau of Meteorology, 2018). The average maximum temperature for summer (December-February) is 25°C. The Melbourne Metropolitan area spans two major bioregions: the Gippsland Plain in the east of the city and the Victorian Volcanic Plain in the west. Bioregions are a landscape-scale approach to classify Victoria’s environment using attributes of climate, geomorphology, geology, soils and vegetation (Victoria State Government, 2018). The Gippsland Plain bioregion is characterized by marine and non-marine Cainozoic sediments and mild temperatures. Mean annual rainfall ranges from 600 – 1100 mm, and daily mean temperature across the bioregion ranges from 9 – 15°C

(Victorian Environmental Assessment, 2010). Much of the vegetation in the region has been disturbed and converted to agricultural land use or (more recently) to urban development. In western Melbourne, the Volcanic Plain bioregion is characterized by Cainozoic volcanic deposits forming a basaltic plain. Mean annual precipitation ranges from 450 – 840 mm, and daily mean temperature across the regions ranges from 12 – 15°C (Victorian Environmental Assessment, 2010). Much of this landscape has been converted to agricultural (grazing) and urban land uses (Royal Botanic Gardens Victoria, 2017). The central and western neighborhoods of the city are generally more industrial than the eastern neighborhoods due to urbanization history.

The community gardens used in the study were selected and stratified based on the criteria that they were allotment gardens in which individuals or households manage their own plots and were representative of the two bioregions. The gardens are managed by individual gardeners or a committee of gardeners, and overseen by the city council government. To control for differences in bioregions, we focused our study area to the inner suburbs that fell within the Victorian Volcanic Plains and Gippsland Plains bioregions (n = 5 in the Volcanic Plain bioregion; n = 6 in the Gippsland Plain bioregion). Moreover, we selected gardens that had 40 -60% impervious surface cover surrounding them (see 2.2.2) to control for potentially confounding urban landscape influences. The gardens are six to 38 years old, are from 584 to 6,801 m^2 in size, and have 25 to 124 allotment plots (Figure 1).

2.2. Temperature variability and plant species richness

We measured ambient temperatures and plant species richness for individual garden plots and for the whole garden. We refer to these two scales of data collection and data analysis as garden scale and plot scale.

2.2.1. Temperature logging

We placed four temperature loggers (Onset HOBO UA-001-08) in each garden to collect hourly ambient temperature measurements (°C) over the summer sample period (December 15 – February 10, 2018). This period is generally when water availability is most limited and temperatures are highest, which are two factors that are associated with high evapotranspiration and stressful conditions for plants in urban environments and warrants targeted research (Faeth et al., 2005). The sample period duration is comparable to other temperature studies in urban agroecosystems (Lin et al., 2018; Piacentini et al., 2014), and was limited by garden access. We worked with garden managers to identify four volunteer gardeners' plots that were spatially distributed within the garden in which to monitor temperature and collect additional plot scale vegetation data. We placed loggers 1.5 m above the plot, and fastened white plastic shields over the loggers to protect loggers from ultraviolet radiation that may damage the sensors and inflate ambient temperatures. We checked and maintained the loggers throughout the survey period to ensure that they were in good working order. Data were downloaded at the end of the survey period and quality checked and cleaned.

For each plot, we calculated the mean and the coefficient of variation (CV) of the average daily temperature. In addition, we calculated the mean temperature CV for each garden from pooled data from the four loggers in each garden.

2.2.2. Plant sampling

We sampled plant species richness and percent cover in each garden across the entire garden (i.e., at the garden scale) and in temperature monitored plots. At the garden scale, we sampled plants within randomly placed 1 x 1 m quadrats along transects placed every 5 m across the garden. Because gardens were of different sizes, we proportionally increased the number of 1 x 1 m quadrats relative to garden size; all gardens had a minimum of eight 1 x 1 m quadrats, and we added one 1 x 1 m quadrat for gardens > 800 m^2 for every additional 500 m^2 (resulting in up to 19 quadrats in the largest garden). We divided the number of quadrats by the number of transects in order to determine how many quadrats to sample along each transect. Within each quadrat, we recorded the species identity of all plants present, estimated the percent cover of plants, and collected information on ground cover characteristics (% grass, straw, mulch, rock, and bare soil) to collect information on local garden characteristics. In addition, we measured the number of trees and shrubs within the garden as a potentially important climate variable (Lin et al., 2018).

Within each monitored plot, we recorded the species identity of all plants present within the entire plot. We also measured the size of the plot, as garden plots were of different sizes. All plant sampling and ground cover surveys were conducted from January 8 – 12, 2018.

To determine landscape-scale plant cover vs urban cover, we collected spatial data of impervious surface cover from Melbourne Water measured at a 1 m spatial resolution (Melbourne Water, 2012) and placed a 1 km buffer around each garden to calculate percent impervious cover within the buffer area.

2.2.3. Analysis

We performed a three-part analysis consisting of multiple models to determine the drivers of temperature variability at the garden scale and plant species richness at both garden and plot scales.

The first analysis at the garden scale aimed to determine whether landscape (regional) or local (garden) factors drive garden temperature variability. Here we used generalized linear models (GLMs) to compare temperature variability among bioregions and among local garden scale factors. We built two sets of garden scale models using pooled data from the four data loggers for each garden. For the first model, the pooled mean temperature CV and the mean daily temperature were the response variables and bioregion was the predictor. We fit the models and ran a post-hoc test using the *glht* function in the *multcomp* package (Hothorn et al., 2008) in the R statistical environment (R Development Core Team, 2016). For the second model, we modeled mean temperature CV and mean temperature by two local scale variables that highly correlated with other local plant and ground cover factors, % grass ground cover (square-root transformed) and garden size (log transformed), and one landscape-scale variable, the % urban impervious surface surrounding the garden (square-root transformed).

The second analysis aimed to evaluate plant species richness at both the garden- and plot scale. We calculated the total number of plant species recorded in each monitored plot and for each garden. To evaluate whether surveys had reached plant species saturation, for the gardens (n = 11) and for the garden quadrats (n = 109), we calculated sample-based species accumulation curves at the genus level for all of the plants recorded in the garden and in the garden 1 x 1 m quadrats. We used "random" sampling methods in the *speccomm* function in the *vegan* package in R (Oksanen, 2015) to generate mean species accumulation curves and the standard deviation calculated from random permutations of the data without replacement (Colwell et al., 2012). A lomolino model was fit to the exact accumulation (Lomolino, 2001). In addition, we plotted species accumulation curves at the garden and quadrat scale using the same method for each bioregion to compare regional biodiversity under the hypothesis that species richness, if influenced by temperature variability, would be different for the different bioregions.

The third analysis aimed to determine whether temperature variability influences plant species richness at the garden scale and at the plot scale. For the garden-scale model, we built GLMs with the following non-correlated variables: total plant species richness observed in the garden (response), mean temperature CV (predictor), and log-transformed garden size (cofactor). For the plot-scale model, we built GLMs with the following variables: observed plant species richness in the plot (response), mean daily temperature CV in the plot (predictor), and log-transformed plot size (cofactor). We built separate GLMs for each bioregion to control for collinearity among explanatory variables and for bioregion. Models assumed a Poisson error distribution appropriate for count data within a given time and space. The best model was selected as the one with the lowest Akaike information criteria (AIC).

2.3. Influences on gardeners' planting decisions and water use behavior

Concurrently with temperature monitoring and plant sampling, we distributed a survey questionnaire to gardeners in all gardens to collect information on gardener decision making and beliefs concerning climate change. The questionnaire asked gardeners about their watering practices, plant selection, their beliefs about climate (temperature, precipitation), and their beliefs about climate in relation to their watering practices and plant selection.

2.3.1. Gardener questionnaire design

We designed the questionnaire to elicit responses on gardening behaviors (water use, planting decisions), and on general beliefs and attitudes that inform gardener decision making. We designed the survey to include multiple choice questions, 5-point Likert scale statements, and one open-ended question. The multiple choice questions asked gardeners what influences how much water they use, and what they add to their soils. A series of 13 5-point Likert questions asked gardeners to indicate how strongly they agree with statements on climate change beliefs, and on the relationship between climate and watering and planting practices. A series of six 5-point Likert questions asked gardeners to indicate how important certain plant species attributes are, including: provision of food/usable products, beauty/aesthetics, cultural meaning, low maintenance, native to Australia, and water use/needs. The one open-ended question asked gardeners to elaborate how the climate patterns over the past 12 months influenced their watering and planting practices.

2.3.2. Participant selection and recruitment

The questionnaire was distributed in an online format by the garden managers to the community garden e-mail list, and was also distributed in paper format by the researchers and garden managers opportunistically during garden work days. The questionnaire was provided in English, and we used professional translators or other garden members to assist with questionnaire distribution for non-English speakers. Gardeners received a pack of seeds in gratitude for their participation. We aimed to get as many gardeners as possible per garden, recognizing that our aim to reach all ~700 gardeners (estimated by reported total gardeners from managers) was limited by language (English) and time constraints.

2.3.3. Questionnaire analysis

Questionnaire data was reviewed, cleaned and quality checked before analysis. We calculated summary statistics for gardener practices and Likert question responses to climate questions and water use and plant-related questions. We qualitatively reviewed responses to the open-ended question of how the climate over the previous 12 months has influenced gardening practices. We performed a thematic analysis of the responses. We first reviewed all responses through which we identified three distinct themes: (1) gardeners stated observations on how they believe the climate is changing but did not provide any information about how climate affects their planting or watering practices; (2) gardeners stated changing or adapting their practices to climate changes and how (i.e., through plant selection or through watering) but did not provide any information about how climate is changing; and (3) gardeners stated both observations on climate changes and how they are changing their practices to these respective changes. We then coded each response with the respective theme: “observational,” “behavior change,” or “observational and behavior change.” For the second theme, we further coded whether gardeners reported on their planting, their watering or both planting and watering.

3. Results

3.1. Temperature variability and cultivated species richness

3.1.1. Landscape and local drivers of garden temperature variability

Mean daily temperatures ranged from 22.2 – 23.2 °C at the garden scale and 21.2 – 26.1 °C at the plot scale, whereas temperature variability values (mean temperature CV) ranged from 22.4 – 31.9 at the garden scale and 19.8 – 35.4 at the plot scale. Although average mean temperatures did not differ between the two regions at the garden or the plot scale (Table 1a), mean temperature CV was significantly greater in gardens in the Gippsland Plain bioregion than gardens in the Volcanic Plain bioregion (Figure 2a). In addition, for garden-scale temperatures, temperature CV was lower in gardens with higher impervious surfaces surrounding them (Figure 2b), and mean temperatures were lower in larger gardens (Table 1b).

3.1.2. Plant species richness at the garden- and plot scale

We observed 655 plants of 122 species across 80 genera from the 11 gardens. Over all gardens and over all sampled 1 x 1 m quadrats, the species accumulation curves did not asymptote to indicate species saturation (Figure 3). Likewise, species accumulation curves did not asymptote individually in the two bioregions at the garden or quadrat scale. Curves for both regions followed similar trajectories. At the garden scale, plant species richness significantly increased with garden size (Table 2). At the plot scale, plant species richness was positively correlated with plot size in gardens in the Victorian Volcanic Plain region but not in the Gippsland Plain (Table 2; Figure 4b).

3.1.3. Influence of temperature variability on plant species richness

There was no relationship between plant richness and temperature variables or bioregion at the garden scale. At the plot scale, plant species richness was negatively correlated with higher temperature CV in gardens in the Gippsland Plain but not in the Victorian Volcanic Plain bioregion (Table 2; Figure 4a).

3.2. Influences on gardeners' planting decisions and water use behavior

The urban climate of Melbourne influences the gardening of the 189 community gardeners we surveyed. We highlight the main results and themes from the survey questionnaire using quotes from the open-ended question and proportions (%) determined from Likert statements and multiple choice questions.

3.2.1. Gardener perceptions of climate in relation to gardening

Overall gardeners described the climate as both variable or as mild over the past year. Some gardeners reported that it was a "mild season" or that "we haven't had drought this year," and some of these gardeners went on to say that the climate has not affected their gardening (11%). However, 61% of gardeners strongly agreed that the climate is changing (in the Likert statements), and gardeners described the climate as unpredictable (in open-ended responses). One gardener described the climate as "increasingly less predictable and less consistent - warm periods when should be cold, and cold periods when should be hot. Everything to excess frequently." Another gardener reported: "Easterly systems moving into Gippsland appear to [be] becoming more frequent and heavier and may reach us. Prolonged periods of higher temperatures in summer. The autumn break seems to be later and less reliable." This gardener went on to share that they believe: "our climate appears to be changing but has yet to settle to a new pattern to meet a warming atmosphere. I feel that in the future it will be less pleasant to live in Melbourne and that traditional vegetable and fruit growing will be forced to change." Thus this gardener links climate changes in the city to broader outcomes for urban life and urban agriculture.

Gardeners are in strong agreement that both natural rainfall and temperature influence the way that garden plants grow (> 80% of gardeners), and tended to agree that they are concerned about the effect of increasing heat (50%) and drought (50%) on their gardens, and that drought will cause water scarcity (60%). Gardeners reported incidences of and worry about plant mortality due to extreme climate events. To illustrate, one gardener shared: "I have been concerned about whether my plants will survive or if I will lose harvest due to severe weather." Other gardeners thought that "the hot days seem harsher on veggies now," and observed "the blistering sun burning/frying foliage." One gardener reported that they "lost all [their] tomatoes with the heat," while another gardener reported that high humidity caused tomato wilt. However, the same gardener stated that "other plants like cucumber have done really well." Thus while some vegetables were reported to fail to grow or produce, others may survive climate extremes.

3.2.1. Gardener watering behavior in relation to climate

Gardeners reported that their water use is most influenced by their beliefs on what plants need (89%) and the climate (76%). Some gardeners also reported that garden rules influence their watering (37%) as well as the soil conditions, water conservation efforts, and time constraints (< 10%). A majority of gardeners agreed that they change the way that they water in response to the climate (50%) and specifically based on the temperature (60%). To the open-ended question, gardeners report changing both the times and the regularity of watering to maintain plants in response to inconsistent climate patterns. Gardeners in this group said that they are: "increasing their watering due to the 40 degree days we had"; "increasing watering due to higher temperatures over longer periods"; and doing "more frequent watering due to increased temperatures." Gardeners reported that they are "more conscientious about watering to keep plants alive." Described by another gardener, "I pay more attention to the weather report and respond quicker to dry and hot weather […] I am more conscientious of our plants' needs." However, other gardeners reported that they have used less water or changed their watering method due to more rainfall over the year: "we have had some extreme rainfall so I haven't watered as much"; "more rainfall so less watering, but heavier watering less frequently"; and, "used less mulch, more subsoil watering."

3.2.3. Gardener plant selection in relation to climate

Only 30% of gardeners agreed that they change the plants that they grow in response to climatic changes in precipitation and temperature (Likert statements), and only 9% of gardeners described how they are changing their planting practices to climate changes (open-ended question). Most gardeners disagreed that past drought experiences influence the plants that they currently plant. Rather, the plant attribute most important for gardeners is the provision of food/usable products (90% of gardeners ranked “important” or “very important”) (Figure 5). This attribute was followed by low maintenance (35%), beauty/aesthetics (33%), water use/needs (30%), habitat for animals/insects (22%), cultural meaning (18%), and native to Australia (9%).

3.2.4. Diverse responses to climatic changes and associated challenges

Few gardeners (4%) agreed that they are changing both planting and watering practices in response to the climate. One of these gardeners reported: “I have not planted any plants which are too temperature sensitive. I try to minimize the amount of water I use in the garden by less frequent but deeper watering, mulching, etc.” Another gardener shared that: “I water more often as I feel the water evaporates more quickly. I also mulch more now to help keep the soil moist. I choose plants that can tolerate harsher conditions. I take care to protect myself from sunburn so I try to garden in the morning or early evening.” Thus this gardener shared how the climate influences their water use behavior and plant selection, as well as when they use the garden. Like this gardener, others reported increased mulching to reduce water loss and watering needs: “I am now purchasing twice the amount of mulch to try to retain more moisture in the soil and moderate soil temperature.” In addition to protecting soils by mulching, gardeners reported adding shade cloth to protect their plants from solar radiation.

Last, some gardeners indicated that learning how to garden in a changing climate is a challenging process. In the words of one gardener: “I find the garden more vulnerable and responds if I fail to care properly for it. I can be hit and miss at times, and feel my routine is not right yet.” Another gardener stated: “It's been unpredictable, making it hard to know when to water […] We've had some very wet periods […] also some very hot periods, which has been a challenge to manage.” One gardener described how plant establishment is increasingly challenging: “Planting new plants requires more watering in and care time to establish. The late heat in late 2017 meant tomatoes didn't establish until much later. Direct sowing is more challenging with less reliable rainfall. We have had to rely more on seedlings.”

4. Discussion

Temperature variability within urban gardens is largely driven by landscape context, and this variability is challenging gardeners to adapt their behaviors. Temperature variability shapes plant species richness at the scale of an individual’s plot, in addition to the area available for them to garden, but this depends on regional context. Gardener reports tell us that they are challenged by, but responding to, the climate changing around them by changing their watering practices and to a lesser extent by changing the plant species they select to grow. Rather, gardeners state they continue to select plants that they think are able to provide food and usable products.

However, the evidence of the species recorded growing in plots shows that temperature variability does influence the plant composition of plots in regions with more temperature variation. within a more variable region, suggesting that plant survival and species distribution is related to temperature and not just people's preferences for plant attributes. Thus although gardeners may be able to overcome some temperature effects by adjusting their watering behavior to maintain the plants they consider important for food provisioning, gardeners are not able to completely mitigate these effects within climatically variable regions and plant species richness continues to be shaped by climatic drivers. In the following discussion, we explore these two related findings from our work: 1) temperature variability and plant species richness in gardens, and 2) their relationship to gardener beliefs, preferences and behaviors.

4.1. Temperature variability and plant species richness in gardens

Landscape context (bioregion) is a significant determinant of community garden temperature variability at both the garden scale and plot scale. Gardens and plots in the Gippsland Plain, a bioregion with more natural land cover, experience significantly larger fluctuations in daily temperatures than gardens in the Victorian Volcanic Plain, a bioregion with greater impervious surface cover. Indeed, gardens surrounded by more impervious surface had lower temperature variability (or higher temperature stability). However, daily temperatures within these regions are similar, suggesting that the temperature fluctuations are more extreme when in natural surroundings. Urban heat island effects could be stabilizing daily temperatures in gardens, as heat is absorbed and retained by impervious surface throughout the day and released slowly in the night to reduce large temperature fluctuations (Grimm et al., 2008; Oke, 1973). Higher temperatures from urban heat islands have been found within similarly structured urban community gardens in comparable urban regions in the USA (Lin et al., 2018). In Melbourne, urban heat island effects within green spaces are documented (Torok et al., 2001) but are neighborhood context dependent (Coutts et al., 2007). Interestingly, at the garden scale, local factors including greater grass ground cover and the number of trees and shrubs did not have strong effects on temperatures in gardens (i.e., were not strong predictors in our models), even though greater grass ground cover (Huang et al., 2008) and tree densities can significantly cool urban green spaces (Berry et al., 2013). This could be because grass (turf) and urban trees are often not irrigated in Melbourne in contrast to other urban environments (e.g., Southwestern USA). Thus if and how local or landscape-scale land cover affects urban garden microclimate may be very dependent on regional context.

A species area relationship is strongly driving urban garden plant species richness at the garden scale, and at the plot scale in regions where temperatures are more stable (Victorian Volcanic Plain). Garden scale plant species richness accumulates with garden size (i.e., a proxy for cultivation area), and size could be weakening the relationship of temperature variability on total species richness on the garden scale, a relationship that we found at the plot scale only in the region where temperatures are more variable. Garden size is correlated with the number of plots and the number of gardeners, suggesting that gardens with more gardeners and therefore more diverse management practices have higher plant species richness (Kendal et al., 2010). We also observed that the larger gardens had more communally managed cultivated areas that are often cared for by a group of gardeners, likely increasing species richness and chances of plant survival. At the plot scale, gardeners may have more species in their plots if they have more space to cultivate and experiment with if they are less challenged by regional temperature variability that could affect more sensitive plant species. This may be due to differences in an individual's management and capacity to maintain high plant richness in climate extremes (discussed below). Plant richness in urban home gardens is similarly positively correlated with increasing garden size (Loram et al., 2008; van Heezik et al., 2013), and home gardens have similar high species richness to these allotment gardens (Clarke, 2014; Loram et al., 2008; Smith et al., 2006) including gardens in the Gippsland Plain bioregion (666 species vs our 655 species) (Threlfall et al., 2016). These studies in home gardens have found comparable species accumulation patterns from sampling efforts to ours. For example, Loram and colleagues (2008) documented > 1,000 species within home gardens across five urban regions in the UK; however, their species accumulation curves were far from saturation after 120 samples. Clarke (2014) found that the total 278 observed species in 104 home gardens in Los Angeles, California, USA did not asymptote, even after extrapolating to 200 gardens. Along with these studies our findings further the argument that urban gardens are diverse in plant species and research has not captured all of the agrobiodiversity that they harbor (Galluzzi et al., 2010).

4.2. Gardener planting decisions and watering behavior

Gardeners in this study strongly agree that temperature and natural precipitation influence the way that garden plants grow, and strongly agree that the climate is changing. The gardeners state that climate changes in temperature and precipitation are challenging their garden management. In response, gardeners often try to mitigate climate extremes by adjusting watering behavior in efforts to support plant survivorship and crop production – gardener's state that they visit their garden more frequently during extreme heat events to provide supplemental irrigation to plants. Although we did not directly measure water use, increased watering frequency suggests that gardeners are likely using more water to maintain their plants based on what they think their plants need. Water use monitoring in community gardens in California, USA found that gardeners water longer and use more water when temperatures are higher (Lin et al., 2018), and gardener water use behavior is similarly influenced by the perceived water requirements of the plants that they are growing. Although gardeners readily adjust their water use behavior to climate changes, the majority of gardeners do not adjust plant selection to climate/temperatures. Rather, gardeners select plants that provision food or usable products: food provisioning was three times more important of a plant attribute to gardeners than water use/needs or other cultural factors. This suggests that plant species in gardens are driven by plant traits associated with food production (e.g., higher flower and fruit set) and ecosystem services rather than plant traits associated with water needs or temperature thresholds. This is similar to reported preferences for ecosystem service-based traits of urban trees (Pataki et al., 2013). However, as mentioned above, we have evidence that plant survival is nevertheless related to climate because plant species richness declined with increasing temperature variability in the garden plots in the bioregion with more variable temperatures. Thus water use behavior may be able to mitigate some temperature effects to maintain plants, but not entirely, and temperature variability can remain as an environmental filter of plant survival if there are strong regional climatic effects on the local climatic context (Williams et al., 2008). A garden in a more variable local climatic context versus one in a more stable local context could have higher chances of plant mortality if gardeners in those gardens are not able to quickly respond to, for example, temperature extremes. Indeed, the high abundance and distribution of "alien" crop species and rare species including ornamentals in home garden systems are explained by gardener management and supplemental irrigation (Loram et al., 2008; Clarke, 2014). In sum, our study suggests that plant species presence and species abundance within urban gardens are mediated by gardener preference for specific plant attributes, ability of gardeners to adjust water use behavior to climate changes to prevent plant mortality, and landscape- and plot-scale variability in temperatures.

5. Conclusion

The relationships between landscape, temperature variability and gardener behavior delivers perspective on the future sustainability and planning of urban gardening. Because reported gardener resource use behavior is largely informed by temperature and precipitation fluctuations and extremes, our results suggest that gardening in more urbanized areas may have some surprising food production benefits for urban gardeners in comparison to the peri-urban fringe (here the Gippsland Plain). More stable temperatures within the urban core regulated by regional drivers may accommodate productive, species diverse and sustainable fruit, flower and vegetable gardening to provision food and well-being benefits provided the space to do so. From an urban sustainability and urban planning viewpoint, as cities like Melbourne densify in structures to meet population growth, urban gardens should be better incorporated into the built fabric of cities through environmental and social reform efforts. Urban gardening can support crop diversity to improve food security, and could have climate mitigation potential in the city (Lovell, 2010; Piacentini et al., 2014). In conclusion, urban gardens are diverse agroecosystems that are shaped by individual gardener management and as well as by landscape-scale environmental factors, and this can likely affect resource use in the city. The relationships among environmental factors, human decision making, biodiversity and subsequent water use should be carefully considered in city climate adaptation plans.

Tables and Figures

Table 7-1. GLM analysis at garden- and gardener plot scale of the relationships between mean temperature (C°) and bioregion, and temperature CV and bioregion (a); best model (lowest AIC score) predicting important local and landscape factors of gardens for garden mean temperatures and temperature CV (b). Victorian Volcanic Plain (VVP) is the reference level in (a); temperature is abbreviated to "temp".

	Scale	**Response**	**Factor**	**Estimate**	**SE**	**z**	**P**	
a.	Garden	Temp CV ~	Bioregion (VVP)	-2.88	1.39	-2.07	0.04	
		Mean C ~	Bioregion (VVP)	-0.09	0.23	-0.40	0.69	
	Plot	Temp CV ~	Bioregion (VVP)	-2.81	1.25	-2.24	0.03	
		Mean C ~	Bioregion (VVP)	-0.14	0.30	-0.48	0.63	
								AIC
b.	Garden	Temp CV ~	(Intercept)	58.63	9.88	5.94	<0.001	49.4
			% Impervious	-4.40	1.40	-3.15	0.01	
		Mean C ~	(Intercept)	24.90	0.88	28.43	<0.001	8.5
			Garden size (log)	-0.63	0.28	-2.27	0.05	

Table 7-2. GLM analysis at garden scale (a) and gardener plot scale (b) of the relationship between plant species richness and temperature CV, and garden size or plot size for each bioregion, respectively.

Scale	Bioregion	Response	Predictor	Est.	SE	z	P	AIC
a.Garden	All	Plant species # ~	(Intercept)	2.04	0.91	2.25	0.02	69.9
			Temp CV	-0.02	0.02	-0.76	0.45	
			Garden size (log)	0.58	0.17	3.49	<0.001	
b. Plot	Gippsland Plain	Plant species # ~	(Intercept)	3.93	0.46	8.45	<0.001	161.6
			Temp CV	-0.05	0.01	-4.45	<0.001	
			Plot Size (log)	0.46	0.26	1.78	0.08	
	VVP	Plant species # ~	(Intercept)	2.46	0.44	5.62	<0.001	117.8
			Temp CV	-0.01	0.01	-0.78	0.44	
			Plot Size (log)	0.60	0.23	2.63	0.01	

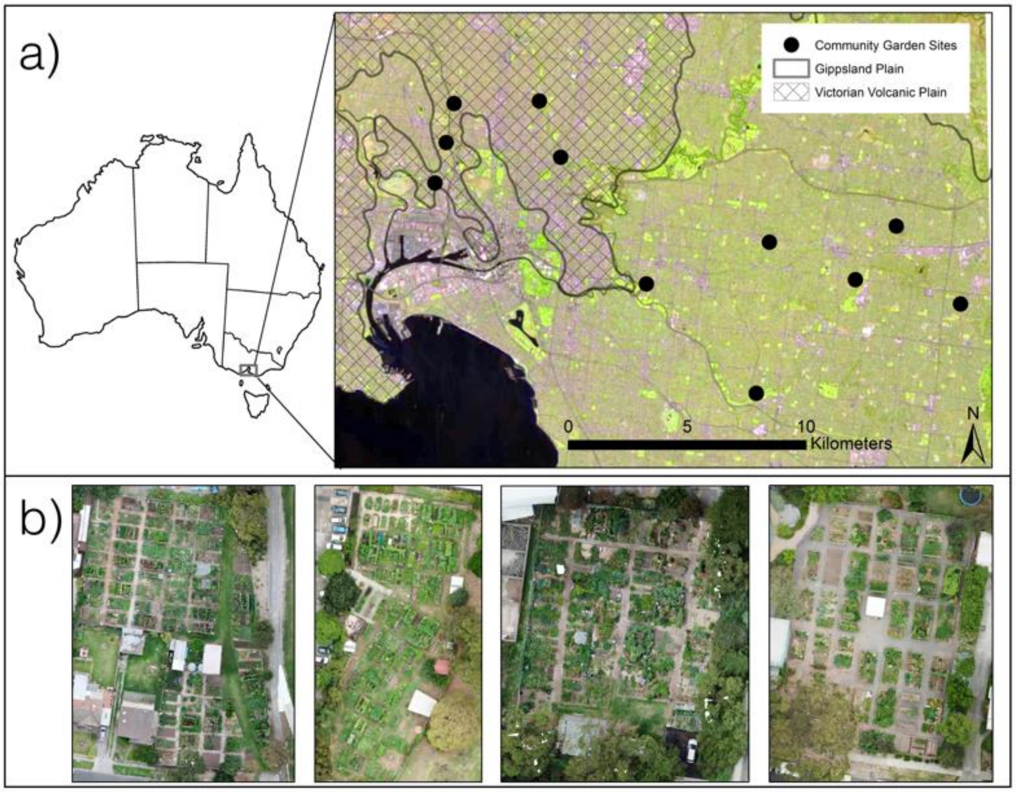

Figure 7-1. Locations of the 11 gardens studied within two bioregions in the Greater Melbourne Metropolitan area in Victoria, Australia (a). Aerial images of four of the studied gardens (b). The fist two on the left are within the Victorian Volcanic Plain, the two on the right are within the Gippsland Plain.

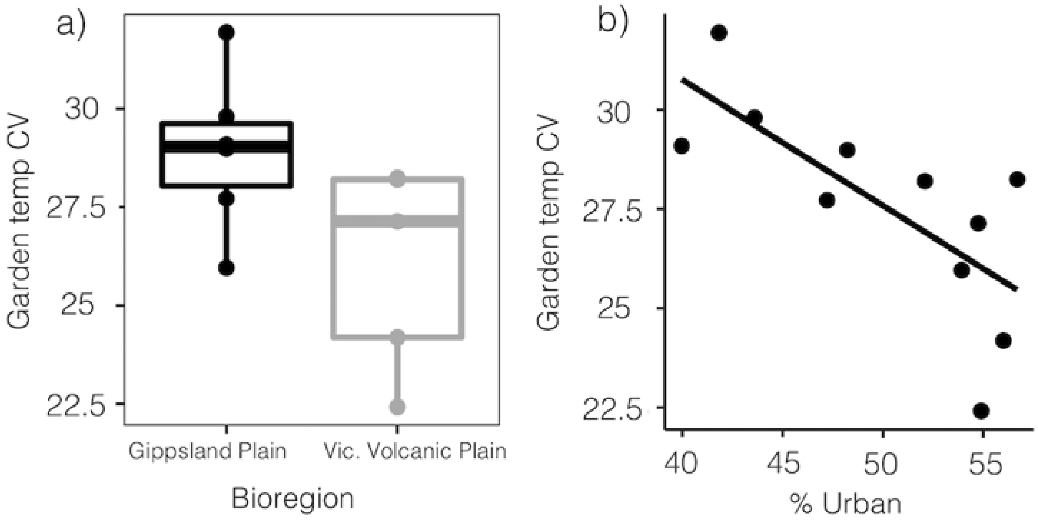

Figure 7-2. Garden ambient temperature variation (CV) was greater in the Gippsland Plain bioregion (a), and in gardens surrounded by less urban impervious surface (%) (b).

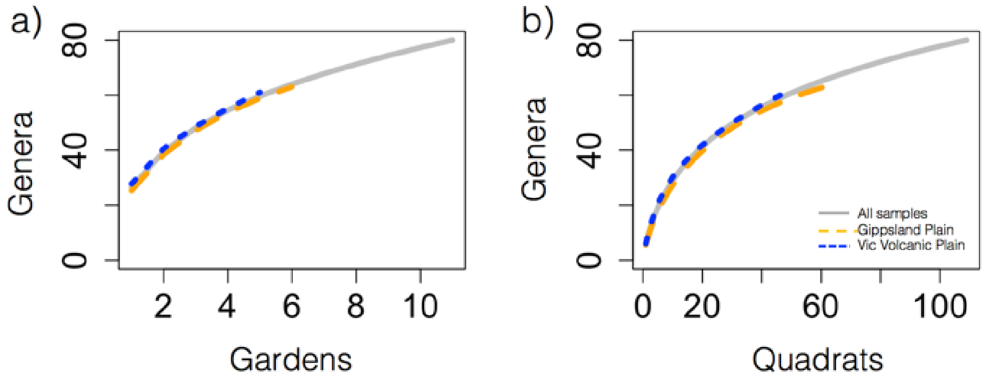

Figure 7-3. Species accumulation curves for genera sampled from the gardens (a) and within the sampled quadrats (b). Gray lines indicate for all pooled samples, large orange dashes represent the Gippsland Plain bioregion, and small blue dots represent the Volcanic Plain bioregion.

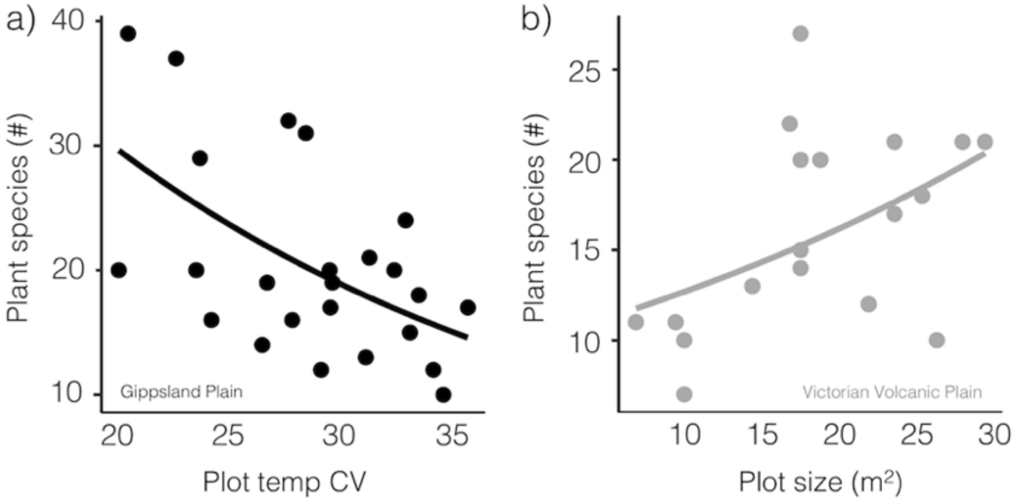

Figure 7-4. Relationship between measured plant species richness within monitored gardener plots and ambient temperature variability (plot temperature CV) in the Gippsland Plain bioregion (a), and gardener plot size (meter squared; log scaled axis) in the Victorian Volcanic Plain (b). Lines are Poisson regressions.

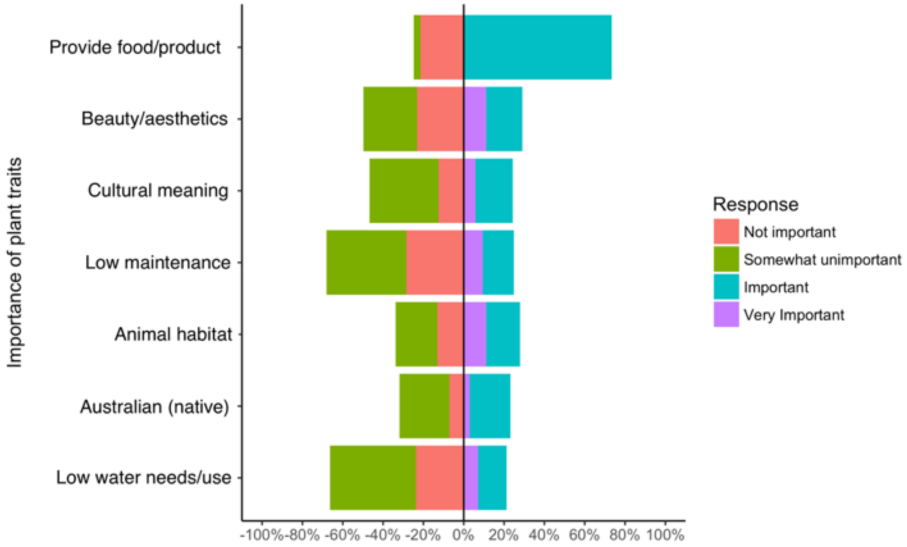

Figure 7-5. Responses to Likert scale questions asking how important plant attributes are to gardeners' plant selection. The y-axis shows plant attributes and the x-axis indicates the percentage of gardeners with a strong directional response. Here, positive values indicate a positive response (i.e., attribute is important), negative values indicate a negative response (i.e., attribute is relatively unimportant), and neutral non-directional responses (i.e., do not agree nor disagree) are at zero.

Part III

8. Gardener wellbeing along social and biophysical landscape gradients

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published in *Sustainability*, 2018

Abstract: Increasing human populations are challenging cities to grow sustainably while maintaining green spaces that deliver ecosystem services and well-being benefits. Community gardens are green spaces that provide food, community, and health benefits, but gardens often are non-permanent due to development and green space loss. Thus, investigating their significance and benefit across urban regions is critical for research and policy alike. This study investigated the role of community gardens in providing human well-being benefits across three counties in the California Central Coast—a region undergoing massive urban transformation in the last century. We measured how multiple aspects of self-reported gardener well-being varied in relation to the social opportunities of surrounding neighborhoods and the biophysical features of the landscapes in which the gardens were embedded. The results document improvements in gardener well-being through gardening across social and biophysical gradients. Gardeners are motivated by diverse reasons, varying from gardening in order to connect to nature, to gardening for improved food access, or to enhance time spent with family. Community gardens are therefore important for supporting many well-being benefits. Policies to maintain and protect gardens should prioritize neighborhoods with needs for connecting to nature and enhancing social interaction within the community.

Keywords: green space; ecosystem services; environmental justice; social opportunity; urban planning

Introduction

Population growth, the increased density of built infrastructure, and the reduction of vegetation cover are extending regional urbanization across landscapes [1]. The decrease of natural habitats in urban areas can result in the deterioration of human well-being due to the loss of ecosystem services, including regulating (e.g., climate regulation), provisioning (e.g., food), and cultural services (e.g., recreation), among others [2]. Urban people often live in environments with low biodiversity, food insecurity, and social alienation due to urbanization. In response, urban planners are carefully considering how to integrate green spaces so that cities can grow sustainably and to support the benefits to society provided by these spaces [3].

Increasing green space availability dually supports the biophysical functioning and sociocultural services of city landscapes to improve environmental sustainability. Green spaces can, for example, reduce surface and air temperatures [4] and thereby reduce urban heat island effects [5]. At the same time, resident exposure to the natural environment through active engagements such as gardening, for example, can reduce the stress and anxiety associated with urban life and provide mental health benefits [6,7]. Moreover, urban green spaces can provide physical health benefits through increased physical activity which can lower the risk of cardiovascular disease [8] and type 2 diabetes [9], and exposure to the outdoors that can also boost immunoregulation [10,11] and lower mortality risk [8]. Therefore, urban green spaces contribute to physical, mental, and social dimensions of human well-being [12].

The multiple benefits of green spaces should make these areas more attractive and amenable to visitation by urban residents, especially in built environments with high levels of impervious cover [13]. Moreover, they may provide an important space for individuals to obtain these benefits who lack access to private green spaces of their own [13,14]. Unfortunately, often urban residents who have little to no access to private green spaces also suffer from a deficit of public green spaces [15]. These residents may be the most disadvantaged in terms of social and economic opportunity within the population [15–19]. Thus, access to urban vegetation is highly influenced by socioeconomic factors [20,21], and improving green space availability and access across social advantage gradients may be the key to promoting both well-being and environmental justice in cities [22].

Community gardens are a popular green space where urban residents grow fruits, vegetables, and flowers either individually in allotments, or in collective schemes [23,24]. Gardens can increase fresh food access [26], support mental and physical health [26–28], and build community cohesion and social networks [29,30] through outdoor recreation. While the social benefits of community gardens are well documented through specific place-based case studies [23,31–33], we still lack information about how community gardens provide well-being benefits to people across urban areas and socio-environmental gradients. We know less about how these gradients influence gardeners' use of, and experience within, community gardens, and the importance of community gardens for green space well-being benefits to users. Moreover, it is not known if time spent in gardens is a function of landscape surroundings or social opportunity. This is necessary information because there are growing concerns that, as with other green spaces, the benefits of community garden access and participation are not equitably distributed among urban residents [34,35]. Depending on social and biophysical neighborhood context, some gardens may be considered 'vital urban spaces' for residents if they indeed provide nearby green space to those who generally lack access [36]. Therefore, it is important for urban policy and planning to consider where and how gardens are allocated and used in city landscapes to promote the benefits associated with gardening as well as equity among communities.

The present study examines garden use and the well-being benefits derived within community gardens along social and biophysical landscape gradients. The goal of the research is to investigate how self-reported garden use and well-being benefits vary in relation to the social opportunities (e.g., in housing, education, and environmental quality) of the neighborhoods and biophysical features (e.g., urban form, and natural land cover) of the landscapes in which the gardens are embedded. Specifically, we investigate whether (1) time spent in gardens is related to neighborhood social opportunity or surrounding landscape biophysical features, and

(2) there is a relationship between the derived well-being benefits from gardens and the neighborhood’s social and biophysical features across multiple study regions.

Materials and Methods

Study System

We conducted this investigation across three counties in the California Central Coast, a region of great biophysical complexity, rapid urban population growth, high levels of human diversity, and substantial levels of socioeconomic inequality [37]. The Central Coast region is increasing in density of built infrastructure to accommodate population growth but remains among the most unaffordable urban areas in the USA [38]. Community gardens are a desired green space for residents with limited access to the natural environment; however, availability can be limited due to long waitlists [39]. We worked in 18 of the region’s urban community allotment gardens during the 2017 summer growing season in the Monterey (36.2400° N, 121.3100° W), Santa Clara (37.3600° N, 121.9700° W), and Santa Cruz (37.0300° N, 122.0100° W) counties, in California, USA (Figure 1). The gardens were 405 to 8134 m^2 in size, 2 to 39 years in age, separated from one another by >2 km, and were surrounded by a mix of natural, agricultural, open green space, and impervious land cover (Figure S1). The 18 gardens were selected because they were all managed in an allotment style where households cultivate individual plots within the garden and because they are relatively well supported by local organizations or by the city government.

The neighborhoods around the gardens varied in terms of biophysical features and sociodemographics due to different histories of urban development and demographic change. Santa Cruz County is a leading producer of strawberries and leafy greens. Yet, many of the workers that pick these fruits and vegetables live in food insecure neighborhoods [40]. Southern Monterey County is known for its maritime industry, US Pacific Naval forces, and tourism, which have brought cultural diversity and economic affluence, as well as socioeconomic hardship, to the region [41]. Santa Clara County has transformed over the past half century from the "Valley of Heart's Delight"—an orchard landscape tended to by Asian and European immigrants—to "Silicon Valley"—an impervious landscape of growing socioeconomic disparity [42]. Across the region, community gardening supports thousands of urban gardeners, many of whom use gardens as an opportunity to grow organic food, be in the outdoors, and build community [39]. Here, we focused on garden use and the well-being benefits derived, as self-reported by a portion of these gardeners in a subset of the region's gardens.

Neighborhood Biophysical and Social Opportunity Data

We used landscape biophysical data from the US Geological Survey's National Land Cover Database (NLCD) [43]. Within 5 km buffers surrounding each garden, we calculated the percentage of land cover types by dividing the area of each type within a buffer by the total area in each buffer using spatial statistics tools in ArcGIS (v. 10.1) [44]. We created four land cover categories: (1) natural land cover (combined deciduous forest [NLCD land cover class # 41, evergreen forest # 42, mixed forest # 43, shrub/scrub # 52, and grassland/herbaceous); (2) urban land cover (combined low built development intensity # 22, medium built development intensity # 23, and high built development intensity # 24); (3) open land cover (developed open green space like parks and recreational spaces # 21); and (4) agriculture land cover (combined pasture/hay # 81, and cultivated crops # 82) (please see [45] for descriptions of land cover classes). The categories describe the main forms of land cover in the region, provide a way to analyze broad landscape-scale patterns, and also incorporate finer-scale urban landscape heterogeneity (i.e., presence of green space) [46].

We collected neighborhood socioeconomic and sociodemographic information for the census tracts surrounding gardens from the Regional Opportunity Index (ROI) [47], an index that assesses the relative well-being of people and places for census tracts in California. The ROI uses data from the American Community Survey [48] and other data sources to create "place domains" calculated from two or more indicators that describe a neighborhood's relative assets in education, the economy, housing, human health/environment, and civic life. For this study, we used five place domains that capture the social characteristics, specifically the social opportunities, of garden neighborhoods: economic, housing, health/environment, education, and civic opportunities (Table 1).

Community Gardener Data

We used survey questionnaires in each of the gardens to collect information about garden use by gardeners and the benefits of gardening to gardeners' well-being, including consumptive and non-consumptive benefits. To measure garden-derived well-being benefits, we used qualitative methods for well-being analyses that may offer further insight into garden-derived benefits [23,26] than numerical measures [49]. Using qualitative semi-structured questions, we asked gardeners if and how community gardening has a positive impact on their or their family's well-being. The question format was open-ended to allow gardeners to elaborate. To measure garden use, we asked gardeners how many hours they spend per week gardening. In addition, to validate the social and spatial scale of the analysis, we asked how far away gardeners live from the garden. We surveyed between 6–14 gardeners per site, which represented between 9.5–65% of the gardener population in a site, the lowest estimated at 10 of 105 gardeners. The surveys were given in English (n = 142), Spanish (n = 38), Korean (n = 1), and Bosnian (n = 1), and were either read out loud by the researcher in person (n = 150) or via phone (n = 2), filled out by the gardener themselves (n = 27), or read out loud to the gardener by another gardener (n = 3).

Data Analysis

For the analysis, we first reviewed all open-ended responses for gardening well-being benefits. Guided by the gardener responses, we then selected codes that were informed by the cultural services defined by the Millennium Ecosystem Service

Assessment [50] as well as peer reviewed literature on the social benefits of community gardening. The codes included food, community, family, physical health, mental health, learning and knowledge, sharing with others, and connecting to nature (Table S1). Because the survey allowed for an open-ended answer, each response could have multiple codes. All responses were able to be coded by the selected codes. Coding was performed using Dedoose qualitative software [51].

We used multivariate techniques to parse out how the variation in responses may be explained by biophysical landscape and social neighborhood features. We used a non-metric multidimensional scaling model (NMDS) to explore how gardens clustered in the relative abundance, or frequency, of coded benefits that they provide to gardeners. Here, the dependent variables were the total number of well-being codes in each category summed for all gardeners sampled from each garden. The explanatory variables were (1) the biophysical features of the landscape, and (2) the ROI social opportunity domains for each garden site. We created three distance matrices to do an ordination analysis using Bray–Curtis dissimilarity for the well-being response matrix, and Euclidian distance for the landscape and neighborhood matrices. First, we used the *vegdist* function and the *metaMDS* function with the *vegan* package [52] in the R computing environment [53] to calculate the Bray–Curtis distance among gardens in the frequency of their well-being responses. Second, we created two matrices, one for biophysical features and one for ROI domains of garden neighborhoods, using the *envfit* function in the *picante* package in R [53]. Well-being data were transformed, scaled, and constrained to two axes with all well-being codes for each site, and then fit with biophysical landscape features and neighborhood social opportunity domains (including all variables). This illustrated where the gardens were situated relative to the well-being benefits reported by gardeners and the biophysical and social opportunity characteristics surrounding them. To determine whether biophysical features and social opportunity were significant drivers of garden clustering, we conducted analysis of variance using distance matrices (ADONIS) tests using the *adonis* function in *vegan* [52]. We conducted separate analyses for the biophysical features that were significantly correlated (i.e., urban, natural, open). In addition, we conducted an Analysis of Similarity (ANOSIM) test using the *anosim* function in *vegan* to determine whether there were statistical differences in responses among gardens in the three counties in the region (Monterey, Santa Clara, and Santa Cruz), fitting county as a categorical predictor variable to the model.

We tabulated the average number of hours that gardeners spent in each garden. We then used generalized linear regression models (GLMs) to examine the relationships between time spent in the garden, urbanization, and ROI domains. The dependent variables were mean hours spent in gardens and the explanatory variables were percent of urban land cover and agricultural land cover at 5 km (square root transformed) and four non-correlated ROI domains (economic, housing, health/environment, and civic engagement). Because the percent of natural and open land cover correlated with urban land cover, we only included the percent of urban land cover in the model. Using a model selection approach based on Akaike's information criterion (AICc), we identified the best model fit with a Gaussian error structure (as all explanatory variables were normally distributed) using the *glmulti* package and function [54] in R. If models did not differ from one another by >2 AIC points, we averaged the models and took the conditional model fit. We confirmed that residuals from the best fit models conformed to conditions of normality using Shapiro–Wilk tests. We visualized significant predictors of garden use with the *visreg* package [55] in R.

Results

A total of 182 gardeners were surveyed in the 18 gardens. Urban land cover around gardens was generally greater in San Jose County than in Monterey and Santa Cruz Counties, which were surrounded by more natural, open (e.g., urban parks and green spaces), and agricultural land cover (Table 2; Table S2). The ROI domains were quite variable across neighborhoods in the counties. Generally, natural and open land cover tended to be associated with higher health and environmental quality (neighborhood healthcare access and health of the environment) and civic life (neighborhood social and political stability) social opportunity domains. Agriculture land cover around gardens tended to be associated with better housing opportunities (neighborhood availability of affordable housing). Urban land cover and education opportunities (neighborhood provision of educational needs and education quality) were not associated with other biophysical or social opportunity gradients, respectively.

Self-Reported Well-Being

Fresh organic food was the most frequently cited contribution to gardener well-being across gardeners (58.2%), followed by mental health (36.3%), connection to nature (23.1%), family (20.9%), physical health (19.8%), hobby (17.0%), personal satisfaction (14.8%), sharing (13.7%), learning and knowledge (12.6%), community (12.1%), and culture (6.6%). The NMDS, constructed from a well-being distance matrix, revealed that well-being responses fell along the gradients in neighborhood biophysical features and social opportunity domains (Figure 2). Reported well-being benefits were neither significantly different between (ADONIS model: F = 0.84, $R^2 = 0.10$, $p = 0.61$) nor within the three counties (ANOSIM model: Global R: 0.04; $p = 0.349$). Biophysical features and social opportunity domains were not statistically significant in the analysis of variance tests for predicting certain well-being outcomes (Table 3); however, we display how well-being benefits varied along social and biophysical landscape gradients in Figure 2 and qualitatively describe general trends.

One major axis along which well-being responses tended to vary was land cover in the surrounding landscape; however, not all types of responses varied. For example, in gardens surrounded by more natural and open (i.e., green space) land cover, gardeners more frequently cited connection to nature as a benefit to their well-being. In the words of the gardeners: “It is very peaceful and relaxing. Gets you back in touch with nature”; and “because I get to be in a magical place surrounded by birds and hawks. To be in nature just to be”. Gardeners in these landscapes also more frequently stated personal satisfaction was an important benefit. As one gardener said: “it has given me an outlet to be productive”. Another gardener further specified: “I like being outside. I make suggestions to others, I like being helpful”. Finally, these gardeners cited mental health benefits associated with nature connection and described gardening as a means to “relax, decompress, and think the day over, almost like walking through the forest—very therapeutic”. In contrast, gardeners in neighborhoods surrounded by more urban land cover in the landscape tended to cite family time and connection in the context of growing food. In the words of gardeners: “spending my time with my children and teaching them gardening”; and, “my older daughter likes to come and help. I like spending time with her and showing her how to garden”. While some gardeners in gardens surrounded by more urban land cover said that gardening is “a part of our family tradition”, others stated that “as a family we have a new found appreciation of the hard work that goes into growing our own food”.

Another major axis along which well-being responses tended to fall was social opportunity. Gardeners in neighborhoods of greater education opportunity tended to cite community benefits and sharing. For example, a gardener in one of these neighborhoods stated: “I’ve met like-minded people here and community members who I wouldn’t have met otherwise”. Another gardener further emphasized the unique social community of community gardens: “Community is the biggest thing—there’s a sense of community that you don’t find in other places”. Gardeners in neighborhoods of greater housing opportunity, but also of more agricultural land cover more frequently cited physical health benefits. For example, one gardener voiced that gardening is “good physical exercise for me, getting out in the fresh air, makes me feel happier”.

Some well-being responses were not associated with biophysical landscape features or social opportunity. In particular, the well-being benefits of food production that were frequently reported by gardeners in all sites did not vary depending on land cover surroundings or neighborhood opportunity. Moreover, many gardeners brought up cultural benefits but were not more or less likely from one type of biophysical landscape or a certain spot along social opportunity gradients. “Talking to people from different ethnicities, learning about culturally appropriate foods”, was expressed by one gardener in this group, while another shared that the garden improves their well-being because they are “surrounded by a nice place with flowers and people from my country”. Cultural benefits tended to be associated with other well-being benefits, like learning and knowledge. As said by one gardener in this group: “I don’t want to lose my habits and ways from my original land”.

Self-Reported Garden Use

The number of hours that gardeners spent in gardens varied with biophysical landscape features and neighborhood social opportunity. The model that best predicted mean garden use included urban land cover and housing opportunity (AICc = 78.72, df = 4). Gardeners spent on average 6.8 h per week at their community gardens, but gardeners spent more time in gardens surrounded by urban land cover (Figure 3a; p = 0.07) and in neighborhoods of lower housing opportunity (Figure 3c; p = 0.004). Gardening hours generally declined in gardens surrounded by more natural land cover (Figure 3b). Yet, gardening hours did not vary in relation to the health/environmental quality of neighborhoods (Figure 3d) despite that differences in well-being benefits reported tended to fall along this axis. A majority of gardeners lived within 1.5 km (42.4%) or within 16 km (86.3%) of their community gardens, affirming the use of 5 km as a buffer in which to analyze landscape features and to use the ROI domains based on US census tract data.

Discussion

Our investigation into community garden use and benefits shows the importance of urban gardens in the provision of well-being benefits across a range of biophysical landscape and social contexts within the Central Coast of California. We show that urban gardens can play different roles for supporting community members' well-being and that garden use also correlates with levels of urbanization in the surrounding landscape as well as housing and environmental availability in the surrounding region. These results support the common view that community gardens are essential urban green spaces in rapidly urbanizing landscapes. They are also important for supporting ecosystem services such as enhanced resident well-being and increased provisioning of food products (i.e., provisioning services) to gardeners [25,56], in addition to a suite of other less tangible cultural services to their users [23,28,32].

Fresh organic food production was stated as the primary benefit of urban gardening (58%) and, unsurprisingly, this was consistent across counties and gardeners as the primary activity of an urban garden. The importance of food for all gardeners in this system is clearly evidenced by food's location in the middle of the ordination plot (Figure 2) and furthers the argument that community gardening can increase fruit and vegetable provision [25,26]. However, a number of other notable benefits were highlighted such as mental health, connection to nature, family, and physical health. Many of these benefits relate to and are being investigated within the current research on the benefits of urban green space [7,57]. Thus, gardeners do not only identify their time in the garden as a time for food production, but they report a variety of ways in which they gain additional benefits. This is evident in the multifaceted ways in which gardeners perceive well-being improvement through their survey responses. Similar to community garden literature in other regions, the responses especially illuminate how these green spaces are important for social relations by facilitating interactions among family and friends [33,58], cultural heritage values by growing culturally significant food plants [59], and educational values by providing a space to share and teach agricultural knowledge [26,60].

Although well-being benefits are not spatially confined to a county, particular types of well-being benefits tend to be situated along social advantage domains and landscape gradients. This suggests that gardens provide specific services to groups of people in relation to their urban experience, and that this in turn may be informed by the social and biophysical amenities that people experience in their neighborhoods. In our study, we show that people who are surrounded by greater urban and less natural land cover tend to spend more time in the gardens, whereas those that have more natural land cover surrounding them tend to spend less time in the gardens. This interesting finding suggests that gardeners' use of garden space is either consciously or unconsciously mediated by the availability of landscape-level natural land cover. Moreover, because more urban areas in this region are also associated with lower social opportunities and therefore likely worse well-being, our findings suggest that gardeners in these areas may have a greater need for well-being benefits and thus use gardens more.

Based on self-identified well-being benefits and garden use behavior, we see that at one end of the spectrum there are gardeners that tend to live in more natural landscapes and choose green spaces like community gardens in which to spend their free time. Many state that they glean benefits from gardens, including personal satisfaction, mental health, and connecting with nature. Community gardening builds relations between humans and nature, develops nature appreciation, and expands environmental awareness [61]. In addition, previous studies also demonstrate that individuals with higher reported connections to nature spend more time in both private and public urban green spaces [62] and that nature orientation can affect multiple aspects of people's green space visitation behavior [15]. Interestingly, however, we found that these gardeners actually reportedly spent less time in these gardens compared to gardens in more urban landscapes. While these gardeners may nonetheless have a higher preference for or access to natural landscapes in which to live and recreate, less reported time in gardens may be because gardeners' time is spread across many public and private green spaces in their surroundings, rather than concentrated within gardens.

At the other end of the spectrum, there are gardeners who live in more urban areas with less access to public green spaces, natural land cover, and private green space. These neighborhoods also generally have less access to social opportunities relative to more natural landscapes, suggesting that these residents also have worse well-being to begin with. In these areas, community gardens are potentially the only access urban residents have to green space and to receiving the ecosystem services and human well-being benefits in a built landscape. It therefore makes sense that gardeners who spend more time in their community gardens also live in areas with more urban cover and less housing opportunity. This indicates that surveyed gardeners are likely more limited in access to urban green space (both public and private) because of fewer and smaller private yards or reduced open green space based on densification and housing costs. Here, gardeners living in these urban areas with relatively poor well-being may have a greater need for the well-being benefits derived from gardens, and consequently visit and spend more time in these gardens. As voiced by one gardener in a garden in a dense urban context, one key benefit to the garden is "access to land now that we live in an apartment—land that we can play and experiment with". Shown here, gardeners in these neighborhoods indicated that they appreciate and use their gardens as a green space (i.e., for its biophysical or natural features) but also a space to connect with family and as spaces for learning and knowledge exchange. This suggests that these spaces are vital and unique for social and community relationships and thus social well-being. This finding supports conclusions that green spaces in urban situations where vegetation and common areas are sparse, are critical for mediating social contact and cohesion among residents through frequent visitation and use [36]. Thus, community gardens in more densely built landscapes with little private green space are likely to be more multifunctional in their uses and diverse in their impacts on well-being.

Gardens are important spaces for community and education. Gardens can provide safe areas for community building and cultural exchange by incorporating cultural events for socializing [63]. Moreover, these are important places to pass on and foster knowledge of ecological systems and skills such as food production to future generations [60,64]. Our research supports these findings, as some gardeners highlighted the importance of these areas for “family” interactions, tradition, and knowledge learning. Krasny and Tidball [60] place learning in community gardens as a form of civic ecology education based on intergenerational community engagement and environmental stewardship. Our results are evidence of these linkages with many gardeners stating that community and sharing are key well-being responses along the education opportunity domain gradient. This suggests that those gardeners that live in more educated neighborhoods use gardens to further develop their own and others’ education through knowledge exchange in a communal environment. These gardeners envision gardens as spaces to share ideas and grow food as a community, and may put more emphasis on—and get more benefits from—the community gardens as a collective endeavor.

In this study, we investigated the self-reported use of community gardens and well-being benefits derived by gardeners from community gardening across biophysical and social opportunity gradients, and show that gardener responses tend to vary in relation to their neighborhood biophysical and social context. Yet our analysis is limited to the scale of the gardener and to the scale of census tract data, meaning that our results may not be generalizable to all gardeners within a garden (particularly where we accessed only a small number of the garden population) or to other populations as we did not use quantitative well-being metrics. Moreover, while a majority of surveys were given orally by the research team (78%), using different techniques (e.g., other gardeners' language assistance) to increase our sampling effort to, for example, non-English speakers may introduce a conservative sampling bias in our results. In addition, we only surveyed garden plot holders to answer how gardeners use and benefit from community gardening, yet future work could access non-gardeners in the neighborhood to explore how garden presence in an area may have spillover benefits by improving the broader community well-being of many other residents [65,66].

Conclusions

The greater California Central Coast region continues to grow in population, increase in the density of structures in the built environment, and is one of the most inequitable urban regions in the United States [38,67]. Urban consolidation and expansion threatens both private and public green space access for urban residents and the ecosystem services they provide [68], especially for socially disadvantaged groups who are already vulnerable to economic pressures [69,70]. Therefore, urban policies and planning need to carefully consider the benefit that green spaces such as community gardens provide to people, especially given our findings that this benefit and overall garden use may be driven by social and biophysical surroundings. Community gardens are high quality green spaces, especially in more urban areas, that provide multiple ecosystem services and have the potential to enhance human well-being across multiple aspects. Policies and planning with an eye for sustaining these benefits need to maintain the presence and longevity of community gardens in cities [71]. This is particularly evident for—but not limited to—those neighborhoods most vulnerable to green space loss where community engagement and connection to nature could be most vital.

Table 8-1. Description of the Regional Opportunity Index (ROI) used for the analysis of the social landscape, the neighborhood attributes that they assess, and the American Community Survey (ACS) data (among other data sources) used as indicators.

Opportunity Domain	Neighborhood Assessment	ACS Indicators
Economy	Neighborhood job and business climate	Job Availability (#), Job Quality (%), Job Growth (%), Bank Accessibility (#), Business Growth (%)
Housing	Neighborhood availability of sufficient and affordable housing	Housing Adequacy (%), Housing Affordability (ratio)
Health/ environmental quality	Neighborhood health care access and health of the environment	Prenatal Care (%), Distance to Supermarket (%), Health Care Availability (#), Air Quality (PM 2.5)
Civic life	Neighborhood social and political stability	Neighborhood Stability (%), US Citizenship (%)
Education	Neighborhood provision of educational needs and education quality	High School Graduation Rate (%), UC/CSU Eligibility (%), Teacher Experience (%), High School Discipline Rate (%)

Table 8-2. Minimum, maximum, and mean with standard deviation values for the biophysical landscape features (at 5 km) and neighborhood social opportunity measured via the ROI across the community garden sites in this study. See Table S2 for all values for each community garden.

Measure	% Natural	% Open	% Urban	% Agri.	Education	Economy	Housing	Health/ Env quality	Civic life
Min	0.0	5.4	12.0	0.0	38.3	10.6	30.5	6.7	69.7
Max	52.3	19.7	94.4	37.0	64.0	21.8	59.1	62.6	90.9
Mean	15.8	12.5	50.2	4.7	51.1	15.5	45.6	38.1	80.0
SD	15.0	4.6	32.0	11.3	7.8	2.7	7.9	20.2	6.8

Table 8-3. Results from the analysis of variance models predicting the well-being matrix. Environmental matrices (social and biophysical) are calculated by Euclidean distance.

Social opportunity	**Df**	**Sum of Sqs**	**Mean Sqs**	**F**	R^2	*p*
Education	1	0.07	0.07	1.06	0.06	0.40
Housing	1	0.05	0.05	0.72	0.04	0.68
Health/Environmental quality	1	0.03	0.02	0.36	0.02	0.92
Civic life	1	0.09	0.09	1.33	0.08	0.26
Biophysical landscape						
Urban	1	0.08	0.08	1.23	0.07	0.27
Agriculture	1	0.12	0.12	1.96	0.11	0.06
Open	1	0.10	0.10	1.57	0.09	0.15
Natural	1	0.08	0.08	1.20	0.07	0.33

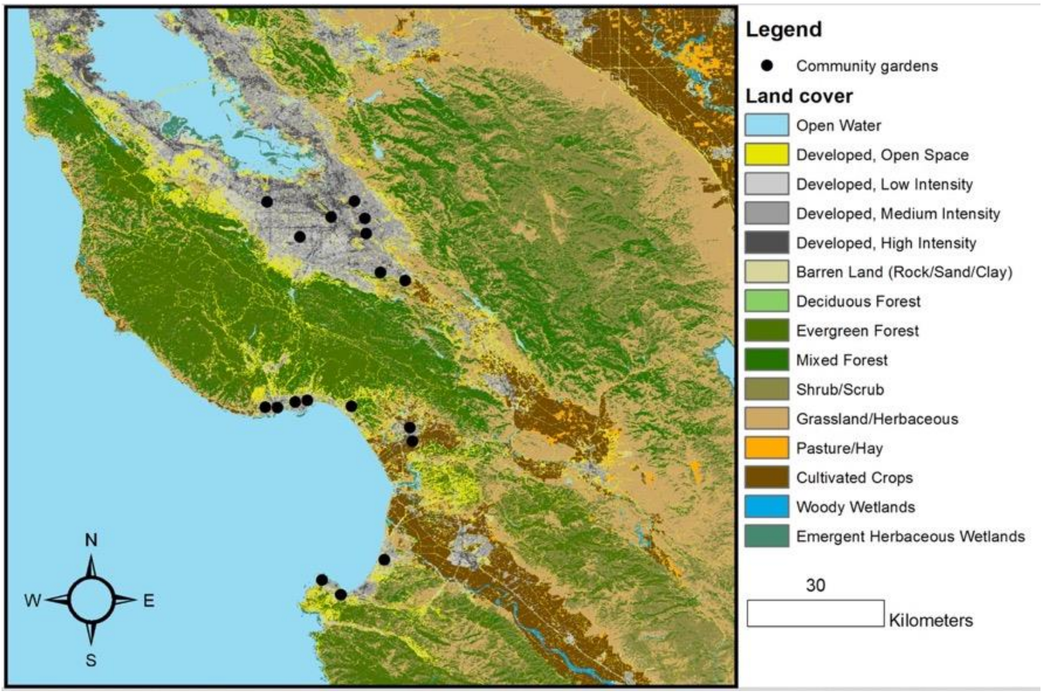

Figure 8-1. The community gardens studied across three counties (Santa Clara, Santa Cruz, and Monterey) in the Central Coast of California.

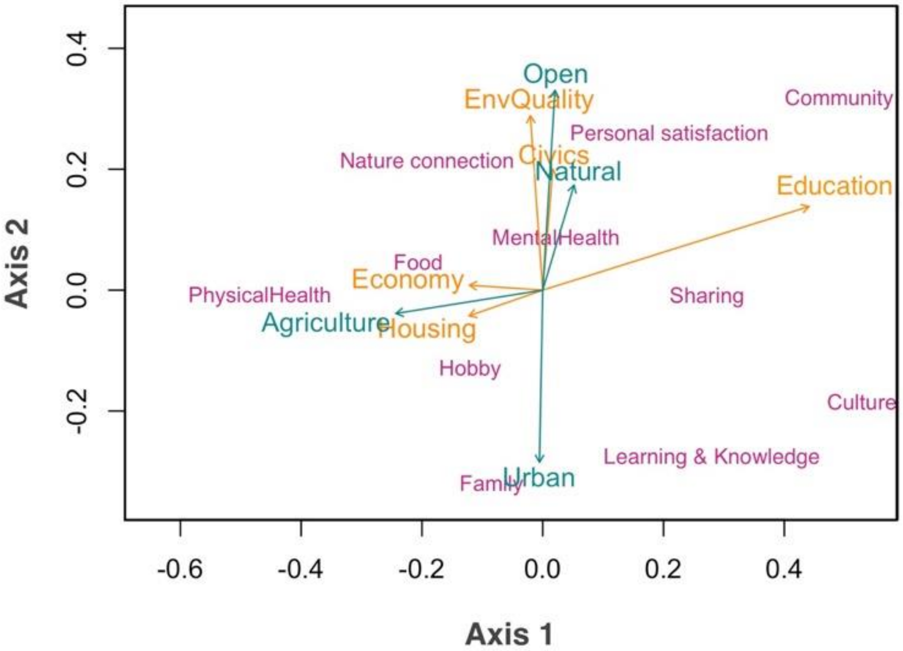

Figure 8-2. A non-metric multidimensional scaling model (NMDS) plot of community gardener self-reported responses to how community gardening impacts their well-being. In pink, well-being benefit responses; in blue, overlaid biophysical features of the landscapes surrounding the gardens (% land cover within 5 km); in orange, overlaid social opportunities (ROI values) of the neighborhoods surrounding the gardens. Biophysical landscape and social opportunity vectors (the arrows) are scaled by their positive correlation to well-being responses, with prediction strength increasing with arrow length.

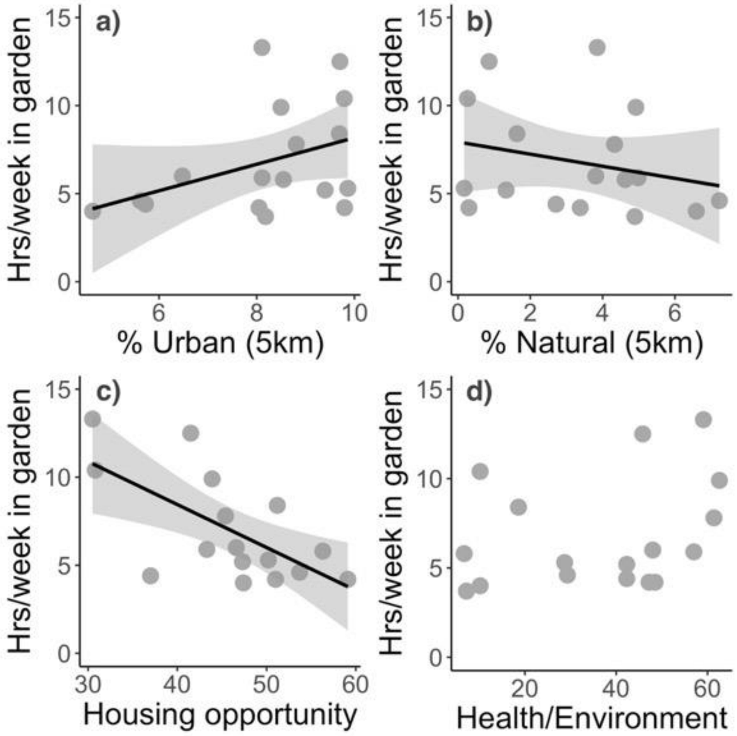

Figure 8-3. Garden use measured by hours per week spent in the garden during summer (May−Oct) in relation to landscape biophysical features (a,b) and neighborhood social opportunity (c,d). Gray circles represent mean values for community garden sites, and black lines represent generalized linear regressions fit to the data distribution. Gray shaded areas are 95% confidence bands. Urban and natural land cover percentages were arcsin transformed to meet assumptions of normality. Greater housing opportunity indicates greater availability of sufficient and affordable housing in a neighborhood. Greater health and environment opportunity describes greater access to healthcare amenities and better health or quality of the physical environment in a neighborhood.

Supplementary Materials: The following are available online at www.mdpi.com/link. Supplementary material will be published online alongside the manuscript, including Figure S1 and Tables S1 and S2. Figure S1: Land cover matrix around community gardens, Table S1: Well-being codes and their descriptors, Table S2: Landscape features and ROI values for all 25 community gardens.

9. Gated gardens: Effects of urbanization on community formation and commons management in community gardens

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published in *Geoforum*, 2018

Abstract

Community gardens are often positioned as spaces where urban people can build community, reclaim common space, and reassert a “right to the city” in urban landscapes that are shaped by gentrification and the privatization of space. However, the literature on urban agriculture often focuses on the struggles of gardens to endure external political-economic processes, largely overlooking within-garden tensions relating to social inequality and resource access. In this study we examined how the pressures associated with urbanization are inscribed in three community garden landscapes in the central coast of California—a region undergoing massive urban transformation in recent decades. The cases reveal that social tensions from urbanization permeate garden boundaries to influence the production of space and the social relations within the garden. Specifically, the resource struggles and social inequities in these regions are made visible in the gardens through conflicts over membership rules, resource management, and theft of produce. The analysis of these conflicts illustrates how extreme real estate valuation and gentrification shapes the particular ways in which the urban commons are managed, including the forms of inclusion and exclusion, claims-making, and racialization of resources that are employed. Uncovering and complicating our understanding of the struggles of and tensions within community gardens is a necessary step in the pursuit of "just sustainability" within changing cityscapes.

Keywords: urban agriculture; enclosure; gentrification; California; resource management; urban sustainability

1. Introduction

"Give me the flowers or I will call the police," Lori shouted, brandishing a pair of garden shears.

"Get the fuck away from me." The middle aged woman she was addressing was undeterred, maintaining her grasp on the bunch of pink roses in one hand and a pair of children's Crayola scissors in the other.

"Give me the flowers, drop the scissors, and I am calling the police. You are not welcome here," Lori insisted.

"Get the fuck away from me lady." The woman shoves Lori, but in the process falls to the ground herself. She drops both flowers and scissors. In what feels like a blink of an eye, she scrambles back up and briskly walks out the the gate down the street, turning into a driveway.

"Hi, I'd like to report an incidence of theft…Yes…I'm at Mayston Community Garden."

This incident—observed during participant observation in a Santa Cruz urban garden—displays a side of community gardening that is not often discussed in contemporary scholarship. Garden shears are not conventionally thought of as weapons, nor roses as sites of neighborhood contestation. Yet, in gardens, where—as one gardener explained—the "worst kind of pest is the two-legged kind," garden shears can take on a completely different role in what (or who) they prune. Fruits, vegetables, and flowers that are cultivated and cared for in community gardens represent more than toil and sweat—they internalize the politics of place within and outside the garden gate. The ways in which gardeners use particular "weapons," from garden shears to personal fences to rules and regulations, reveal the nuanced strategies and practices by which they proclaim a right to community membership, a right to common resources, and a right to space in the city.

Urban community gardens are situated in landscapes where capitalist urbanization transforms nature and social relations (Williams, 1973; Harvey, 1989). Urbanization can broadly be described by land conversion into impervious cover, and by distinct socioeconomic and sociopolitical processes (Grimm et al., 2008). Urban political ecologists (e.g., Heynen et al., 2006a) characterize urbanization processes by: capital accumulation and the externalization of nature (*sensu* Marx, 1976; Harvey, 1983); uneven geographic (both physical, socioeconomic) development (Smith, 1982); and the exclusion and marginalization of some social groups for the benefit of others (Swyngedouw and Heynen, 2003; Swyngedouw and Kaika, 2000). Capitalist urbanization is thus a socio-environmental process of political and economic changes based around material production and exchange that transform humanity's relationship to nature to produce the distinct spatial form and social relations of urban landscapes (cities) (Swyngedouw and Kaika 2000; Heynen et al, 2006a; Angelo, 2016). Specifically, because capitalist urbanization processes tend to emphasize difference in socioeconomic status, unequal power relations between social groups are woven into the social and political fabric of cities (Heynen et al., 2006a). The enclosure of common city spaces in pursuit of capital accumulation frequently results in dispossession and marginalization of underprivileged groups (Harvey, 2003; De Angelis, 2003; Sevilla-Buitrago, 2013, 2015). Cities may, for instance, perpetuate racialized inequality through land use planning and policies that privilege high income homeowners at the expense of low income minorities (Barraclough, 2009). Furthermore, gentrification processes of capital investment and displacement of the poor by new affluent classes frequently reorder neighborhood socioeconomics and demographics (Lees et al., 2013; Slater, 2011). Urbanization, in short, is a pervasive spatial and social process of changing land use and shifting property and power relations (Brenner and Schmid, 2003) that shapes social life (Angelo, 2016).

Urban land transformation does not go uncontested, however. Urban green spaces such as community gardens—collective or allotment style managed spaces for fruit, vegetable and flower cultivation—can be central sites for urban residents to reclaim the urban environment by carving out common spaces and new forms of community (Schmelzkopf, 1995; Von Hassell, 2002; Saldivar-Tanaka and Krasny, 2004; Kingley and Townsend, 2006; Rosol, 2010). In community gardens, residents can grow food and reconnect with nature in a social environment in the context of biodiversity loss, food insecurity and social alienation due to urbanization (Okvat and Zautra, 2011; Guitart et al., 2012). Geographers often theoretically situate gardeners' claiming of space, commons, and natural resources within Henri Lefebvre's (1991, 1996) theories of urban space, including the "right to the city" (see Barron, 2016 for a thorough summary). Here, commoning—collective community ownership and land management—can be “a mechanism for redistribution through which underprivileged residents compensate themselves for uneven urban development” (Eizenberg, 2012: 779). Through commoning, it is argued, community gardening can challenge neoliberal property regimes of urban environments (Blomley, 2005, 2004) and provide residents an opportunity to resist privatization and engage in political discourse and community governance for a more just urban society (Follmann and Viehoff, 2015).

The struggles of community gardens to persist and maintain their commons in the city is captured in a rich literature (e.g., Barraclough, 2009; Irazabal and Punja, 2009; Schmelzkopf, 2002; Staeheli, 2008), but this work tends to focus on the tensions between gardens and external political-economic processes while largely overlooking the within-garden tensions that result from surrounding urban pressures. Depictions of the “community garden” as a singular actor faced with urban stressors can obscure the nuanced ways in which those stressors infiltrate commons management within gardens and differentially shape the garden experiences of various social groups. Some limited scholarship has examined how gardens create community through enclosure or by playing on racial and ethnic difference, thus producing exclusionary spaces that belie idealized notions of community garden inclusivity (Kurtz, 2001; Glover, 2004; Tan and Neo, 2009; Beilin and Hunter, 2011; Ghose and Pettygrove, 2014; Neo and Chua, 2017). Some work has also situated city commons establishment, governance and maintenance within place-based urban political economies (Rosol, 2010) and has revealed how community garden social networks are entangled in uneven urban landscapes of social and political inequality (Domene and Sauri, 2007; Milbourne, 2012). Concerns about garden persistence in a changing political and biophysical context, it has been shown, can shape both community garden internal governance processes and external representation and relations with the city (Gröning, 2005; Spilková, 2017; Spilková and Vágner, 2017). This literature enriches our understanding of community gardens by exploring them through different social perspectives and geographic scales, but it touches only tangentially on the multi-layered and intersectional ways in which urban pressures are internalized within the gardens themselves.

This article explores the varied ways in which city-scale urbanization processes manifest within urban gardens, altering how garden communities are delineated and how communal resources are managed. We draw from fieldwork on the social life of three community gardens in two rapidly gentrifying urban regions in California’s Silicon Valley. We conducted semi-structured interviews with gardeners and garden managers at each of the gardens (approximately 10 at each, 32 total) in the summer of 2017 to ask gardeners about the benefits, challenges, and nuances of community garden participation. In addition, we used participant observation at garden events (e.g., public community events, barbeques and potlucks), and reviewed each of the garden’s rules and regulations to gain insight into community dynamics and to better understand the governance structures of the gardens. To protect participant confidentiality, all garden and gardener names are pseudonyms.

Our analysis situates gardens within their respective complex gentrifying city landscapes, furthering the argument that broader urban dynamics such as racialized othering (Glover, 2004) and enclosure (Neo and Chua, 2017) can complicate urban agriculture's commons management and thus its potential to achieve food justice and enact the "right to the city" (McClintock, 2017). We draw on an urban political ecology framework that necessitates discussions of power, race, and unequal control of resources in the socio-ecological arrangements of cities (Heynen et al, 2006a), as well as research on the conflict and exclusion often entailed in commons governance (Dietz et al, 2003; De Angelis, 2010). In the three gardens we studied, the urban stressors which pervade the Silicon Valley region manifested themselves particularly in tensions over the boundaries of the garden community and over access to the garden's common resources. The configurations of community and of commons management that emerge in each garden as a strategy for coping with these tensions are quite different; facing similar challenges, these gardens react differently. In the garden we call Grovesdale, the need to demonstrate the garden's value in the face of mounting commercial pressures on real estate has led to an exclusive membership regime and the removal of problematic common trees to make room for new member plots. In this case, both community-building and commons management occurs as a top-down process, with power centralized in the hands of garden managers. In Arborway Community Garden, tensions emerge over management of common land and water resources, giving rise to community fissures along lines of race and ethnicity. This case illuminates the racialization of space and natural resources within the garden that muddles portrayals by gardeners (and some scholars e.g., Baker, 2004; Lyson, 2004) of community gardens as oases of biodiversity and human diversity. Finally, in Mayston Community Garden, the social inequality surrounding the garden is internalized through heated battles with non-garden members over theft of garden produce. In this case, community building is a grassroots process, but one centered around the exclusion of non-members.

Our exploration of these three cases reveals that community gardens are not simply a bulwark against growing inequality and the privatization of urban space; these tensions also manifest *inside* the gardens through multi-layered conflicts over such seemingly mundane topics as membership rules, water rationing, and theft of produce. Through our analysis of the complicated lives of these gardens, we illustrate what Staeheli (2008), building on Foucault (1982) calls the “agonisms”—the struggles and reciprocal adversarial interactions among members—embedded in community delineation and commons management with the idea that recognizing these agonisms is an important first step in the pursuit of just sustainability within rapidly changing cities.

2. Urbanization and community gardening in the California Central Coast

2.1. San Jose: “This was once the Valley of Heart’s Delight”

Sprawling between two mountain ranges exists a modern metropolis with a rural agricultural history. California's Santa Clara County was once known as the "Valley of Heart's Delight" for its orchard dominated landscape. The Valley's fertile soils provided fodder for stone fruit, apple, and nut trees, and catalyzed a migration of people to tend the trees and cultivate the land in the later 19th and early 20th century. Japanese, Italians, Portuguese Azoreans, Filipinos and many other populations settled in the Valley to shape the immigrant landscape and ethnic heterogeneity of the city. In the last 50 years, the Valley has undergone a rapid transformation into "Silicon Valley." The physical transformation into an urban landscape has paralleled a sociodemographic transformation into a still highly diverse, yet also highly inequitable social landscape along gradients of race and class (Schafran, 2013). A contemporary migration has occurred in tandem with the technology industry boom. Indians, Western Europeans, and Southeast Asians increased in population numbers in the region as they arrived to work for Hewlett-Packard, Yahoo!, Google and Apple Inc. Rising land values in response to tech wealth in addition to public policy have resulted in regional gentrification (Schafran, 2013)—processes of neighborhood development and change that lead to displacement and marginalization (Hackworth, 2002).

Different parts of the city have experienced changes differently. On the West side of the city, rural ranches, orchards and vineyards have developed into sought-after suburbs with renowned public school systems and multi-million dollar homes that conveniently neighbor technology firms. Home values on the West side are among the most expensive in the country (LOA, 2015; Yelimeli, 2018). Little of the historical agricultural landscape remains—or the farmers, rancheros and farmworkers who once populated it. In the Southeast Side of the city, historically lower income urban neighborhoods including Little Portugal and Japantown are among neighborhoods most at risk for displacement by gentrification in San Jose (Wadsworth, 2017). These neighborhoods still retain their cultural heritage and historical sociodemographics. Cash only mom and pop cafés still serve bar-style espresso and $1.25 *Bolashas De Espece* (horseshoe-shaped Portuguese cookies). Hand-made tofu shops and century old Buddhist churches remain embedded in the cultural fabric of the city. Yet the coming decade is predicted to change this with the scramble to develop condos, townhouses, and apartments to house a growing highly educated, highly skilled new creative class of tech industry workers (Bain, 2014; Nagourney and Dougherty, 2017).

In this changing region, urban agriculture is an increasingly popular pastime for city residents of all demographics. San Jose’s Parks and Recreation places the contemporary pastime in the city’s urbanization history: “since 1977, San Jose has provided an avenue for residents to cultivate their own vegetables, fruits, herbs and flowers, amidst an urban city environment. The Community Garden Program provides a means for gardeners to learn different growing methods from one another, while cultivating friendships, as well as produce. Over 1000 gardeners cultivate nearly 22 acres of community gardens in the ‘Valley of Heart’s Delight’” (Parks, Recreation & Neighborhood Services, 2017). Through urban agriculture, the city clings to its agrarian identity despite its rapid transformation into a concrete landscape.

Here we examine two community gardens—one in the West of the city and one in the Southeast. In the West, Grovesdale Community Garden is a wealthy and ethnically diverse community garden and neighborhood with technology firms just blocks down the street. The gardeners represent the demographic shift of the past 10 to 20 years. As the garden manager describes the changing garden: “Culturally, it was probably very white, Anglo-Saxon when it started, it is now very diverse. We've got Chinese, Vietnamese, Thai, South Asian, Turkish, Russian, Romanian.” In the Southeast, Arborway Community Garden is used by middle and working class participants. It is also very ethnically diverse, with gardeners identifying with Cambodia, Vietnam, Mexico, Poland, the Philippines and other national origins.

The two gardens face a similar set of challenges: urbanization, resource scarcity, and gentrification in a city where demand for housing and land to cultivate is high. Yet the way in which these pressures affect the two local garden communities differs. In Grovesdale, facing land insecurity due to urban revitalization, the management defensively creates high barriers to garden membership to maintain community order. Here, community is carefully constructed and maintained through strict control over access to membership, reminding us that social community formation is not always organic (Staeheli, 2008). In Arborway, facing resource restrictions from city mandates, gardeners draw on racialized representations of resource use in negotiating access to and use of the environmental commons. Here, resource struggles and contrasting philosophies of communal space and management are imbued with the racial dimensions of urbanization in a city shaped by historical racial and ethnic inequality (Schafran, 2013). Though social conflict is inherent to commons management (Dietz et al., 2003; Baud and Dhanalakshmi, 2007), these cases reveal how regional factors shape the *contours* of such conflict and the garden management decisions that result.

2.1.1. Exclusive garden citizenship (Grovesdale)

Becoming a member of Grovesdale Community Garden requires considerable time and commitment. In addition to a wait list over 50 households long (which can take years to reach the top of), prospective gardeners must show city residency, take several two to three hour courses on "organic gardening 101," and take another two to three hour course on sustainable water management. In these courses, prospective gardeners learn the rules, the language, the codes, and the practices associated with garden membership. They learn how to compost and mulch their soils, conserve water, and weed in and around their plots. They commit to hours of commons maintenance and community workday participation. If they pass the courses, gardeners sign a year-long contract for their plot. At the end of each membership year, gardeners are evaluated and must be invited to renew their plots. The long and stringent membership process creates high barriers to entry for any prospective garden members. In interviews, gardeners explained that getting a garden plot generally took several years and the courses were time consuming.

For the managers of Grovesdale, the extensive coursework and maintenance commitments required for membership are a way to cope with the existential threat posed to the garden by gentrification. In the courses, prospective members learn how to become what a manager defined as a “good community gardener,” who follows rules, respects and cares for the commons, and conforms to certain agricultural practices. In their words:

> You can't have something like this without having rules. Our lease is with the city…We pay nothing for this land, so we have to show that we are an asset to the community… Our obligation to the city is to run it well. Not let it fall into disrepair and all of that. Make sure that people are using it wisely…From my point of view, it's common sense. We're a community garden. Yes, you pay for a bed. Yes, you are here to grow, but you also have to participate as part of the garden to help maintain it.

The ideologies of community gardening and the associated rules put in place to uphold them are defended on the grounds of maintaining land and resource access within a city whose revitalization plans for the area threaten garden longevity. Controlled access to garden participation and a highly managed commons can be interpreted as strategies to foster community cohesion (Kurtz, 2001) and produce a functioning garden based on responsibility (Neo and Chuo, 2017). While these are no doubt motivating factors, Grovesdale’s governance structures are also partially a response to the looming threat of dispossession that results from a regional landscape context in which real estate is at a premium. The West side of San Jose has the highest rate of year-over-year change estimates in median housing prices: real estate reports show that land values range from three to four million dollars per acre, with an annual rise of 50% (Bain, 2014), and have experienced a 33.6% change in median asking price for homes (Yelimeli, 2018). In this context, the garden's access to land is necessarily precarious. To maintain its foothold, the garden management tightly enforces rules and micro-manages gardening philosophy in a way calculated to maintain the kind of stable community and evidently productive garden favored by the city.

Some gardeners explicitly understand the need for top-down rules, regulations, and community creation for the sake of community garden endurance in a gentrifying city. Summarized by one gardener, Susan, who serves as a garden board member:

> I don't think you can ever just relax. With this particular garden, because it is city property and it's worth a lot of money, we have to keep it looking really good and [the manager] has been really good about that… [The manager] has had some work groups and she's figured out ways to get people to sign up for those…I think we're a little different from most community gardens. Just the whole layout and keeping the place looking really good. We feel like we have to, to make sure the city's happy with us and feels this is worth keeping.

Susan rationalizes the strict rules and mandated community work that maintain an orderly and aesthetically pleasing garden as necessary for amiable city relations. For Susan, gardener organizing and community building is a challenging feat accomplished from above.

While some perceive order, rules and regulations as necessary for future-proofing the garden, others cringe at the top-down nature of these regulations. As one gardener, Camila, explained: “It's always in a way been a functional dictatorship…Most of the gardeners just come to our gardens…see what's going on but don't necessarily participate more than that…How do I say this? It's called an organic garden but I don't think it always functions organically.” Her description of the garden as a “functional dictatorship” reveals a frustration with the lack of community representation in garden decision-making around commons management.

Compounding frustration with this model of commons management and community-building-from above, the management has recently restructured community governance by assigning “leaders” to a handful of garden sections in order to mediate and simplify all top-down communication. A resulting decline in gardener participatory governance has changed the way in which some perceive and practice their membership in the community and role in commons governance. Illustrated by a gardener, Lily’s, progressive ambivalence: “I don’t know what’s going on with this garden right now because they used to send out certain e-mails but now … it [only] goes to the [section] leader…They make the decision here…Before they used to have meetings for the whole garden, but now it’s only leaders.” When asked why the change in garden governance occurred, Lily simply stated: “I don’t want to get involved in the politics of the garden. It’s easier not to get involved. To garden and then to leave.” The top-down creation of a community of “good gardeners” through mandatory coursework and hierarchical garden governance has thus far been effective at protecting the garden from the external pressures of gentrification, but it also has its discontents.

The garden’s rigid rules for membership and imposed governance regime fracture the egalitarian image of community gardens sometimes depicted in the civic agriculture literature (Lyson, 2004) and complicate the relationship between official membership and a meaningful community. The rule regime helps separate the wheat from the chaff; the “good” from the “bad” community gardeners. Gardeners who do not follow the rules or who are caught for bad behavior (e.g., do not fulfill service hours, “pilfer” from others’ gardens, or let weeds grow around paths) are labeled by management as “bad community gardeners” that do not properly care for the commons and are not invited back to the garden the next year. Managers defend these policies by arguing that peoples’ membership is not renewed only as a result of their own failure to commit to good garden citizenship and commons management and should therefore be considered a mutual parting of ways. However, this relatively rosy depiction of the garden expulsion process is belied by Susan and others who complain of former garden members using their old keys to access and steal produce from garden plots and common areas.

In general, theft or what some gardeners term “pilferage” of produce by both former and current members is one of the greatest threats to the manufactured social fabric of the garden. In interviews, members of the garden management cited pilferage of produce as a “major internal challenge,” increasing in frequency and intensity in recent years despite the locked fence surrounding the garden’s perimeter. Pilferage is particularly problematic within the common areas. In response, the management has defensively installed video cameras and has recently posted signs with explicit quotas for the common trees ("Enjoy one or two pieces, but please, it's for everybody to enjoy"). Managers have even expanded the garden contract to include quantitative restrictions on how much gardeners can take from the common areas. In explaining the pilferage problems, a garden manager additionally pointed to the ongoing need to cultivate community: “Apparently the garden was quite social before…A reason why we probably have theft is people don't know who their fellow gardeners are. We need a way to get people together face to face.” Here community is described as a way to enforce the norms of garden citizenship, suggesting that community formation and cohesion are being mindfully manufactured for specific utilitarian ends. The top-down approach to community formation and commons management taken at Grovesdale aligns with arguments that community gardens produce modes of governmentality (Pudup, 2008) that restrict civic participation (Tan and Neo, 2009) and impart particular expectations and community responsibilities of their participants to uphold a “good” garden and “good” community (Neo and Chua, 2017). At Grovesdale, the techniques used to govern the space and the community have proven relatively effective at preserving the garden's value to a rapidly gentrifying city and at protecting its common resources, but the somewhat autocratic management style creates social fissures and alienates some potential community members.

The pressures of urban gentrification on commons management regimes also played out in the removal of a subsection of the same problematic trees to make way for additional private plots for new members. The garden recently removed its "food forest" of communal fruit trees to accommodate more plots. The manager argues that this was necessary to provide more city residents with access to the garden: "Are we here to grow trees because a couple of people like them or are we here to provide many more people a space to garden? We are a community garden." However, many of the garden's founding members like Camila challenge these transformations:

> I think that the garden is beginning to look like a suburb rather than a garden, especially with trees being taken out…That's what I call the urbanization. I lived in this Valley [since] the fifties. This entire area south of [here] was gorgeous orchards and highly fertile land. It's now asphalt build. That was the basis of comparison that I started using with where I see the garden going.

Camila directly connects physical land use changes in the region to within garden socioecological transformations. She further argued that plot-building is profit driven and suggested that eliminating the common trees calms conflict among garden members by reducing issues of pilferage and over-harvesting of the commons. The manager, however, sees increasing gardener participation as a metric that can be leveraged to maintain the lease with the city by showing that they are an "asset to the community." The transformation of the common areas into plots represents a complex issue. The garden management is, in effect, working towards being more inclusive by adding more plots and opportunities for garden participation (much needed in a place where a home balcony often constitutes the only outside access), yet doing so is simultaneously a means by which the garden management responds to the threat of gentrification by validating the garden's existence to the city. Here, again, surrounding gentrification shapes the production of space within the garden.

The garden, like the city around it, is becoming more dense, reflecting a landscape in which both land to *live* and land to *cultivate* are highly valued (and highly priced). Despite the socio-spatial restructuring and within-garden land use intensification, the garden maintains a degree of social community cohesion through its exclusivity and rigid rules and philosophies. Yet the environmental commons shrink in the process as a conflict-avoidance strategy to improve social community dynamics and therefore maintain a community image to the city. These socio-spatial transformations can be viewed as city-building processes (Brenner, 2016; Brenner and Schmid, 2003) in which the garden accumulates social value by being one of the only green spaces in a built landscape for select residents, as well as by touting its regulated community cohesion. The intensifying land use and conversion of common space to increase gardener capacity are representative of moments of urbanization (Brenner and Theodore, 2002; Peck et al., 2009). Some "old-timers," seeing the transformation and the loss of the trees that were a ghost of the city's agrarian history and its attendant environmental commons imaginary, are leaving the garden, turning their plots over to a changing city.

2.2.2. Racialized debates over commons access (Arborway)

Miles east along a riparian corridor that was once orchard agriculture sits Arborway Community Garden, home to around 60 household plots. The garden manager boasts that, because of its landscape context, Arborway hosts considerable wildlife diversity, including spiders, foxes, skunks, hawks, woodpeckers, and many species of pollinators. In addition to biodiversity, the manager proudly advertises the garden's ethnic diversity:

> It's the most diverse garden in the whole [city] system…We have the largest Cambodian gardeners of any garden. We have either the first or the second largest of Chinese. Right now we don't have any Vietnamese. I do have one African American way down there. Pretty nice guy. Quiet. Sticks to himself. I do have a Chinese family over here, neither the wife or the husband speak English.

However, the human diversity at Arborway complicates social community dynamics due to difference in practice, value, and perception in a physically small space (~1 acre). Further, ongoing regional population growth and housing development, coupled with the regional effects of climate change, is challenging resource access and use in the garden. In contrast to Grovesdale, a lack of top-down control over garden access and greater gardener freedom in management contributes to frictions among community members, often expressed through racialized narratives about resource use.

As with the shared fruit trees at Grovesdale, much of the social friction at Arborway revolves around the commons—in this case water usage and land management. Tensions emerge over how to maintain a socially cohesive, diverse community under stressed environmental conditions. The city has implemented regulations and restrictions on outdoor water use in the aftermath of a five-year drought in California. Gardeners can only water two days a week, Wednesdays and Saturdays, before 9 AM and after 5 PM. This has been challenging for gardeners whose jobs prevent them from making those days or times. Despite strict restrictions, the garden's water bill has tripled over the last three years, a sign that the current mechanisms to reduce water access are not effective for water conservation. The water crisis provokes accusations of irresponsible water use between gardeners. As one elderly white gardener, Dave, described:

> It's a bunch of individuals coming in with individual plots and doing their individual things. If some people overwater and others underwater, then there can be kinds of problems. The garden as a whole, they look at the size of my plot and add it all up and divide it all up and so I pay the same water bill as someone whose overwatering or growing cactus…[Furthermore] If you ration something, that means that even if I don't need to water on Wednesday, if it's going to be hot

> on Thursday and Friday, I'm going to have to come over and water and I'm going to hoard the water because I don't know when I'm going to get it next.

Dave's description updates Hardin's (1968) oversimplified "Tragedy of the Commons" for the modern community garden context, in which the individual plot is prioritized. In his telling, the individual is incentivized to optimize their use of the common water resources when accessible; there is no incentive for gardeners to reduce their water consumption and most gardeners do not consider water a common resource to be conserved for the benefit of all.

Moreover, although the management highlights the garden's ethnic diversity, water sometimes becomes a site of everyday racial politics. Gardeners of different ethnic groups blame other groups in the garden for abusing the water. Some Caucasian gardeners blame Asian gardeners for prolonged overhead spraying; some Asian gardeners blame Mexican gardeners for flooding trenches between the corn. Shortly after Dave described the economics of water use, he argued that corn should be outlawed due to its intensive water requirements and cheapness in the store, overlooking the cultural value of rare corn varieties to Mexican community members. Ethnically diverse community gardens have challenges related to different perceptions of resource management (Kurtz, 2001) and access (Glover, 2004), and here at Arborway we see that these tensions manifest in water use.

Water conservation and water use policing have become the manager's greatest crusade in the face of a ballooning water bill. Printed signs with block letters and the management's signature are taped to sign posts around the garden to remind gardeners of the watering rules. However, the manager also brings race and ethnic difference into his narrative about water misuse:

> I wish that there was a way to monitor every garden, how much water they use. Like the second gardener in from here, he doesn't speak too much English, he speaks Chinese. And he'll stand on one end of his garden and water the other and I told him, "no. You can't do that. No overhead spraying." He just shakes his head like this [shakes head].

At Arborway, the city's water restrictions and the manager's surveillance of water use create an environment in which ethnic diversity among participants becomes associated with water wasting practices. Gardeners of different ethnic and cultural backgrounds manage gardens according to different values and levels of engagement (Head et al., 2004), which can influence the attitudes and practices of water use in gardens (Head and Muir, 2007). However, the public nature of community gardens means that water use practices are conspicuous, in contrast to the hidden water networks of urban domestic spaces (Kaika, 2005). This visible water use is judged by community gardeners in ways that reveal a racialization of commons management.

Like water, land management and values can become politicized and racialized. Fences with locked gates have increasingly been erected around individual plots in recent years. As in Grovesdale, some gardeners attribute this to the problem of intra-garden produce theft, however for others it is explained by the different perceptions of land and resources held by gardeners of diverse ethnicities. Internal fencing has social consequences, creating physical and social divides within the community. Fencing raises visible questions about what it means to be a community within a community garden by exerting a strong notion of individuality that contradicts the common image of community gardens as facilitators of social connectedness (Kurtz, 2001). From the analysis of two long-standing gardeners at Arborway:

> Beatrice: This is a community garden but at the same time it's a little territorial because people are putting up their fences, and it sends a different meaning. It says keep out.
>
> Antonio: It's a community garden but at the same time it loses flare, concept, philosophy of community. Once you start erecting fences, it's "my plot, my land." That's why I just leave it like this. Because you know what, people take stuff, it's no biggie for me because we're old school. We're only borrowing this from the land…But all these fences… you know, it doesn't even belong to you. We just pay the rent. And this guy right here, he's very competitive…He's very territorial. He's from Michoacán…I don't know if you know anything about Mexico, but there's some states where people are very territorial when it comes to land. But I'm not going to put a fence, buy wire. For what? You know it's about a philosophy. It's about a value.

As in the case of water use, ethnic difference among gardeners become weaponized in the management of natural resources. Unlike at Grovesdale, which boasts a similarly high level of ethnic diversity, here the influence of ethnic difference on agricultural practices and philosophy generates differences and potential conflict. In the absence of top-down filtering mechanisms that select for obedient gardeners, and management strategies like courses on sustainable water use and gardening 101 that homogenize gardener practices, philosophies and behaviors clash and are racialized.

The effect of spatial practices like internal fencing on community cohesion within ethnically diverse urban communities highlight how different perceptions held by different groups of people create possibilities for variation in meanings of *community* and *community garden* (Kurtz, 2001). The downside of much-lauded ethnic diversity within community gardens is that it can contribute to conflicting interpretations of best commons management practices and conflicting convictions about the obligation of community gardens to be communal and inclusive spaces.

However, internal tensions around garden resource management and the philosophy of community can also be understood as a reaction to inequitable resource access and territorial arrangements that are intrinsic qualities of urban space (Brenner and Schmid, 2003). Capitalist urban development drives the privatization of limited natural resources and the reorganization of space outside of the garden, and these city-scale socio-spatial processes affect how gardeners claim and enclose resources within the garden. The urban environment is also, in this case, one of racial injustice. The greater San Jose and San Francisco Bay Area are built on racial inequality and the dispossession of some for the profit of others (Brechin, 2006), and more minorities groups experience poverty (Hispanic or Latino (31%) and Asians (24%); US Census Bureau, 2014). These are thus landscapes where past marginalization lives on through contemporary land use change (gentrification) that variegates social and economic opportunity on gradients of race and class, shaping inequitable resource distribution in times of scarcity. Simply, these are landscapes where “new racial geographies and the geography of crisis can be seen at multiple scales” (Schafran, 2013: 678). Low income groups and people of color continue to be disproportionately burdened by environmental contamination and toxins (Dillon, 2013; McClintock, 2015) and by displacement through the fiscalization of land use, redevelopment and gentrification (e.g., Chapple and Strategic Economics, 1999; Dillon, 2011; Schafran, 2013).

In Arborway, city-wide racial tensions are made present within the garden through the racialization of resource use and territorial arrangements that enmesh ethnic difference. Racialized claims allow gardeners to exploit existing social distinctions to deflect blame from their own use of common resources and to focus scrutiny on the practices of others, underscoring the mutual constitution of ideas about natural resource use and racial difference (Moore et al., 2003). This internal friction over commons management differs from the top-down control of commons and intensive community curation exhibited at Grovesdale, but both reveal how regional socioeconomic inequalities can intersect in the micro-politics of these community spaces.

2.3. Santa Cruz: “More crowded, less hippy”

At the lip of California's Monterey Bay lies an increasingly less "sleepy" and more crowded beach town known for its tourist attractions, natural beauty, and region-wide gentrification. The boom of technological service industries on the other side of the Santa Cruz mountains—"over the hill" as locals affectionately term the greater San Jose metro area—has attracted high income earners who opt to live at the oceanfront and telecommute or undertake a daily commute to and from Silicon Valley. This trend has raised land values, leading to skyrocketing housing costs and the displacement of low-income residents. As of 2017, the city is in an affordable housing crisis in which nearly 70% of surveyed residents experience rent burden (spending more than 50% of income on rent), 50% of renters have experienced a forced move, and 27% experience overcrowding (McKay and Greenberg, 2017). The city has the highest rate of homelessness and is, according to some measures, the least-affordable small city in the nation (Out of Reach Report, 2017). Consequently, the beach town historically acclaimed for its "weirdness" and hippy inclusivity is becoming more exclusive, its benefits increasingly out of reach for low income residents.

Mayston Community Garden is a long-standing fixture of Santa Cruz. Most gardeners live within walking or biking distance and spend afternoons and weekends tending to flowers and food crops for recreation and connection to nature. While the garden is secure in its relationship with the city, the gardeners face stressors and threats from other outside groups, predominately surrounding neighborhood residents who would like to partake in the garden's bounty. Mayston demonstrates how the threat of produce and gardening tool theft by outsiders can result in conflicting perceptions of and claims-making to the garden as a managed commons for gardeners' benefit versus as an unmanaged common pool resource for all neighborhood residents to enjoy. Here, gentrification and increasing socio-economic inequality in the city tends to exacerbate tensions between gardeners and outsiders around the issue of theft. Although different in its proximate landscape context from the other two gardens (e.g., lower housing density), this case further demonstrates how capitalist modes of production of space are articulated within the garden through enclosure, here to protect against outsiders.

2.3.1. Threats of theft (Mayston)

Mayston, overseen by the city, consists of 45 households, a mix of longtime residents, many (more than half) of whom are retired or self-employed, middle class, food secure, and white (Authors, unpublished data). Gardeners pay an annual fee of $100 a year for their plots, and the wait list for the garden consists of over 20 people – similarly reflecting the popularity of gardening and limited number of gardens in the area to Grovesdale. The garden is physically connected to multiple neighborhoods because of its unique placement at the end of a cul-de-sac; household fences around the garden's perimeter host gates into the garden to provide access points into various neighborhoods. The connectedness offers benefits to the general public: anyone can access the garden, walk through it with purpose or for leisure, and appreciate its agrobiodiversity. Yet, the garden sends confusing messages to the passerby. At one gate, the sign “no trespassing” is nailed to a thigh-high gate, but at the other end, the garden has a 10-meter-wide permanent opening and a public playground.

The physical context of the garden troubles the garden’s membership. Some gardeners are conflicted about how the garden should function in the social-biophysical fabric of the neighborhood, and how it should interact with the public. Some gardeners reported appreciation for the awe and inspiration that their gardens bring to visitors, stating—in the words of one gardener, Kim—that people are “so surprised to see everything so beautiful and all the flowers.” However, some gardeners reported frustration with the outsiders who consider the garden a common pool resource—rather than a managed and regulated common space—in which the vegetables and flowers are fair game for all.

The biggest challenge that the gardeners face is theft of garden produce and gardening tools from the public; three of the gardeners described people who “come in with their shopping bags” to pilfer garden plots as though they were grocery aisles. All of the gardeners interviewed reported having things stolen: just-ripened vegetables, bags of fertilizer, garden chairs, and, in one report, entire plants along with their cages taken from their plots. Three of these gardeners have reported incidents of theft to the police department, as in the rose theft confrontation that began the article, and most gardeners portray theft as deliberate, disrespectful, and immoral. Half of the gardeners interviewed explicitly critiqued a common perception held by the public that a community garden is a garden *for* the community. As one gardener, Rosie, stated: “It's unclear whether sometimes when they are taking food whether they think community garden means it's for the community, because I've heard people thought that's what it was. When they're stealing tools and digging out plants it's pretty clear they're just stealing." Such gardener narratives pose outsiders as exploiting the perception of community gardens and ignoring gardener expenditures of capital, labor, and time. This is further encapsulated by Margaret:

> I find there are parents that come through with their young children and they just let them into my neighbor's garden to pick raspberries. When I've said something, they say, "Well, there's so many raspberries. They won't even know that a few are gone." It's a strange type of morality how they accept that…I think that when people see food growing or even flowers, that they don't really understand how much energy goes into it. There have been people who have said, "God gave this food to us." They don't realize, yeah, but I've been out here everyday for the past three months working my butt off.

Gardeners like Margaret claim their exclusive right to the space and to the garden's bounty by virtue of their labor as well as the money they lay out for membership. They insist that access to the garden belongs to them, it is a managed commons, not an unregulated common pool resource.

Similar to Arborway, Mayston gardeners have increasingly put up fences around their plots, locking them up when they leave. Yet in this case the fences are a protective strategy to defend against outsiders rather than a means to convey something to insiders; fences at Mayston are erected to say as a gardener Virginia put it: “we know that you are here. Could you please have respect and stop? This garden belongs to somebody. It is not yours.”

Defensive strategies against outsiders have been met with mixed perceptions. Some gardeners say that gardeners “aren’t fence people,” while others fully embrace the use of chicken wire and chains, even calling for video cameras as necessary additional solutions to the chronic theft problem. With the city taking a hands-off approach to the issue, seeing produce theft as the gardeners’ problem, some of the gardeners have collectivity acted to protect and control their space using an informal “neighborhood watch” system in which gardeners police one another’s plots when there is a passerby. One gardener, Will, interrupted our interview three times within 20 minutes to question visitors’ intentions, in one instance responding to an inquiry of “may I walk around?” with “you just can’t pick anything.”

These gardeners’ efforts to protect their space from external encroachment extends beyond the issue of illicit harvesting; some gardeners also object to people using the garden space for sleep and other forms of sanctuary. One gardener reported finding that a young homeless woman had set up a camp in her garden plot for over a week. Disappearing during the day, the woman would return at night to find sleep among the garden plot’s dahlias, kale and carrots. Objecting to incursions from the homeless, the gardener has now blockaded her little garden with a meter and a half-high fence and bolted lock.

Compared to the other two gardens, Mayston maintains a relatively close-knit community of gardeners, but it partially arises and is maintained by gardeners uniting over protecting their environmental commons from produce theft and use of space by outsiders. Conflict with the garden's resources illustrates confusion over unmanaged "common pool resources" and a managed commons (McKean and Ostrom, 1995) in the urban environment. While previous work mostly concerns gardeners fighting the city for land in the context of capitalist urbanization (e.g., Schmelzkopf, 1995, 2002), this case demonstrates that the gardeners may be the ones acting on the side of enclosure—not in the literal sense of creating private property from commons, but through prioritization of individual access rights and physical bounding of space. Enclosure within gardens defines access and inclusion to garden benefits (Kurtz, 2001; Tan and Neo, 2009; Neo and Chua, 2017) and muddles representations of gardens as an urban commons that are non-commodified, open recreational social spaces operated through collaboration rather than competition (Eizenberg, 2012; Hardt and Negri, 2000; Lyson, 2004). Rather, fencing of the community garden conveys "both a symbolic and material sign of seemingly public but actually private and exclusive space" (Neo and Chua, 2017: 676). At Mayston the commons management elicits within-garden community bonding and communication (similar to Neo and Chua's findings), and the gardeners reinforce a public-private dichotomy (Blomley, 2004) through their protectiveness of space. For many gardeners, the garden is a regulated communally managed resource that should only be accessed by dues-paying members. For many outsiders, it appears, the garden is a *community* resource, a common pool resource, in which the flowers and berries grow for all.

In interviews, few gardeners made the connection between chronic theft of produce, homeless incursions, and the regional gentrification that has increased regional inequality. For example, the same gardener who complained about theft and supported installing video cameras also proclaimed gardens as key components of a "healthy" city; in his words: "A healthy community has gardens for people who don't have their own land to do it. This is just a part of the big picture of what [the city] is." Community gardens are carriers of culture within cities (Eizenberg, 2012); their physical composition (vegetation composition, built infrastructure) and social dynamic (social relations, interactions, community cohesion) embody the everyday experiences, practices, and perceptions of garden users. Mayston garden does express certain aspects of the city's culture (e.g., its rich history of sustainable agriculture), but its "healthiness" is questionable. The Mayston case illuminates that, in a city with great social inequality in the form of housing opportunity, the garden tends to rationalize commons access for those who can pay and maximum benefits of access for those who can enclose, protect, and reconfigure the meaning of community and reorganize commons access for their personal use (De Angelis, 2003; Tan and Neo, 2009; Neo and Chua, 2017).

3. Conclusion

Urban political ecologists have thoroughly documented the conflicts that arise between urban green space protection and the imperatives of capitalist development (Irazabal and Punja, 2009; Pierce et al., 2016; Schmelzkopf, 2002, 1995), but the socioeconomic inequality and racial tensions associated with capitalist urbanization can also infiltrate such urban oases, shaping social relations and resource use in fundamental ways (Angelo, 2016). This paper explored the internal dynamics of community gardens in two rapidly gentrifying and increasingly inequitable urban landscapes to better understand how urban political-economic logics inform mechanisms of access and claims-making that shape the political ecology of gardens. We showed that garden communities react to these processes differently, as evidenced by the contrasting management styles and social dynamics in each garden case study. Exploring intensive processes for gaining garden membership, racialized tensions surrounding access to common resources, and conflicts that arise over theft by outsiders illuminated how city-wide racial and economic divisions manifest as intra-community friction over garden resource governance. Although commons management always involves degrees of exclusion and conflict, we conclude that the larger urban landscape of extreme real estate pressure, racial inequality, and resource scarcity shapes community formation and commons managements strategies within urban gardens. Further, the multiple coping strategies deployed in each case study garden reveals that negotiating regional pressures is a plural and non-linear process. While others have explored the question of garden inclusivity (Glover, 2004; Neo and Chua, 2017; Tan and Neo, 2009), our analysis centers the regional landscape context which shape the contours of inclusivity and the tensions that arise within these communities.

Highlighting internal tensions, however, does not entirely negate the societal value of urban gardens. All of the gardeners we interviewed also described their gardens as sources of organic food, community and a connection to the environment. Indeed, learning to manage commons with a diverse group of co-gardeners may constitute a valuable experience in and of itself. As one Santa Cruz gardener described [her] experience: "it's just a beautiful common ground thing… you can connect with people that otherwise you have nothing in common with, but your common ground is this." In Arborway, despite racial tensions and issues of individuality, the garden collectively donated over 200 pounds of produce to a local soup kitchen. It also bestows a plot for a local Girl Scout and Boy Scout troupe to learn how to care for, cultivate their own fruits and vegetables. In Grovesdale, though the process to become a gardener is exclusionary, the garden is publicly accessible during weekly open house hours for the greater community to enjoy, and it often hosts public workshops on topics like composting and fruit tree pruning. Thus there is cause for optimism; though community gardens internalize the social tensions at work in the greater urban landscape, they also positively shape that landscape by incorporating social functions (Lovell and Taylor, 2013; Hou, 2017) that increase their longevity and social sustainability (McClintock and Simpson, 2017), and they may even help foster broader community cohesion (Kingsley et al., 2009; Alaimo et al., 2010) and cross cultural understanding (Hou, 2017).

Unequal cityscapes shape gardens, but gardens also foster new individuals who populate and attempt to change that city (White, 2010; Follmann and Viehoff, 2015; Hou, 2017). As a gardener in our study, Nora, from Arborway encapsulated: “Gardeners come out and you hear them share their stories and they change. So you meet people at the same time and you change…The learning and the interaction with people, and interacting with the environment…When I look about an urban space, I always think about those ideas that other people have tried… If we could only coordinate that information and experiences.” For gardeners like Nora, community gardens are spaces of personal reinvention, but also spaces for experimentation with ideas that can creatively improve the experience of city living. Gardens internalize urban problems but they also bring together ideas on how to respond to those problems, which can be projected back out onto the greater urban environment. Thus the very conflicts over resources and commons management that we have described through the case studies can themselves be learning tools. Through these processes community gardens help gardeners “produce and transform their own urban worlds through everyday practices, discourses and struggles, leading to the formation not only of new urban spatial configurations, but of new visions of the potentials being produced and claimed through their activities” (Brenner and Schmid, 2015:65). In sum, community gardens are intersectional spaces that embody both alienating and commons-building processes simultaneously through their contradictions in community formation and community maintenance (Staeheli, 2008; Tan and Neo, 2009; Neo and Chua, 2017). Though frequently riven by discord, they nonetheless remain realms of possibility and re-imagination.

In sympathy with gardeners who have lost the fruits of their time and labor to what they perceive as theft, and in agreement that water conservation is a pressing issue in urban agriculture, our objective is to show the tensions and "agonism embedded in community, rather than gloss over community's potential or its problems" (Staeheli, 2008: 7). The beneficiaries of gardens are not random, but are filtered through sociopolitical and racial sieves that regulate access to garden communities and environmental commons (Glover, 2004). As a result, gardens are not isolated from urbanization processes, but internalize city-scale injustices and marginalization within the gardenscape's socio-spatial relations (Angelo, 2016). As one gardener stated, community gardens are "a microcosm of the world. So, it brings out some not so good things in people, like greed and entitlement. But it also brings out good stuff: generosity." Channeling this optimism, we invite the question of how community gardens can be more egalitarian as gardeners navigate difference and disagreement over commons management. We see it as critical to unpack and address these struggles in the city in order to foster just sustainability.

10. Creating socioecological novelty in urban agroecosystems from the ground up

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published in *BioScience,* 2018

Abstract

The contemporary renaissance of urban agriculture is crucially linking ecological and social systems in modern cityscapes. Community gardens contribute ecological (biodiversity conservation) and social (food security) benefits to urban residents, and enhance landscape multifunctionality. Cities have been recently labeled “novel ecosystems” due to the multi-scalar interaction of socio-cultural and biophysical elements. We conceptualize community garden soils within the novel ecosystem framework, arguing that gardeners are cultivating novel agro-ecosystems starting from their garden bed’s soil. While urban soil characteristics are linked to land-use legacies in most urban ecosystems, we find that in our system – the California central coast – community garden soils may live outside the influence of legacies associated with geographic location. Instead, the intensive (re)working of soils by gardeners may washout legacy signals and decadal chronological trends, leading to soil homogenization that reduces the biophysical ties to the meters of earth below. Thus, socio-ecological novelty can be created in multiple forms in cities.

Keywords: anthropogenesis; biotic homogenization; socio-ecological systems; urban agriculture

A major change in modern socio-ecological systems is the dramatic shift toward urbanization. Cities concentrate people and resources into relatively small areas, yet form global economies of scale that accelerate innovation and wealth creation while increasing demands for urban land (Batty 2008). This concentration of people within the landscape engenders novel interactions between socio-cultural and biophysical components of urban ecosystems (Alberti et al. 2003), which can generate novel features (e.g., biophysical conditions, species composition) within these communities (Kowarik 2011).

Such interactions highlight the need to better understand the effect that the built environment and associated social systems have on ecosystems, their structure, and their processes. Urbanization often leads to a reduction in vegetation cover within and around cities, as well as a general degradation of remnant habitats. Urbanization can alter biotic communities due to increased fragmentation and impervious surface (McKinney 2002), both of which are landscape features that select for "urban adapted" species (McKinney 2006). Yet, urbanization may also lead to a boost in species richness in comparison to natural areas due to an increase in exotic species planted in residential areas (Threlfall et al. 2016). Together, urban ecosystems are novel within and across trophic levels, and this novelty has implications for abiotic and biotic processes that affect ecosystem services.

Recent evidence suggests that urban ecosystems converge in their structure and function, potentially leading to the homogenization of biotic communities living therein. Urban soils in particular exhibit biotic homogenization and converge across urban ecosystems, in which biogenic properties (e.g., soil carbon and nitrogen) are similar across urban landscapes (Pouyat et al. 2015). Homogenization can be driven by human landscape preferences, norms, and values, through which socio-demographically similar urban areas produce similar biophysical characteristics (Groffman et al. 2014). The replacement of natural cover with turf grass is a prominent example of how urban neighbourhood ecology homogenizes or converges due to social norms (Robbins 2012); here, urban ecological systems are sites of capital accumulation and the intersection of social-economic institutions and the environment (Kaika 2005). Urban ecosystem homogenization, convergence and socio-economic dimensions contribute to the novelty of cities.

Contextualizing novelty

The novel ecosystem framework describes a system as novel in its species composition, interactions and functions (Hobbs et al. 2006). Conservationists and restorationists classify novel systems by the amount of characteristics or conditions retained from historical ranges (Hobbs et al. 2009); novel systems are considered to be completely and irreversibly altered or transformed in species composition and function from their “historical” configurations. While the notions of “historical” and “natural” states are arguable when one considers indigenous land-use legacies (Heckenberger et al. 2003), such baselines have been used as a way to measure ecosystem “divergence” in the wild, the agricultural (Lindenmayer et al. 2008), and in the constructed urban landscape matrix (Pickett et al. 2001). Urban land-use transformation is a driver of biological homogenization (McKinney 2006), creating socio-ecological systems and landscapes that are distinct from those that they replace but relatively similar to one another in structure and functions (Groffman et al. 2014).

In response to homogenization, urban researchers and practitioners have been thinking carefully about how to create and manage green spaces that support native species diversity, ecological processes (e.g., stormwater infiltration, decomposition) and ecosystem services under an *ecology for the city* framework (Grove et al. 2016). One type of green space that has experienced a renaissance in recent years and has piqued the interest of local governments and urban planners is urban agriculture. Urban agro-ecosystems like allotment community gardens (henceforth 'community gardens') are pockets of green space in cities where species diversity, ecological processes, and ecosystem services, and functions may be preserved through strong social-ecological interactions. Gardeners carefully select crops for food, ornamental, and medicinal purposes (Baker 2004), but they also heavily cultivate the ground underneath through soil selection, amendments, and irrigation that supports the planned and associated biodiversity of these systems.

In the following dialogue, we argue that the consistent and deliberate human management that occurs *from the ground up* in community gardens represents a specific example of how novel agro-ecosystems emerge within cities through soil biotic homogenization. We look specifically at community gardens as an emergent agro-ecosystem to show how intensive human management of soils to sustain crop production is creating novel types of ecosystems that are similar across time and space. We suggest that human preference, intensive management and similar soil formation processes (from purchased inputs to knowledge capital) are leading to the homogenization of garden bed soils across urban regional hubs. This supports the idea that social factors, not just environmental conditions, lead to biotic homogenization, ecosystem convergence, and environmental similarity. While most work has focused on urban vegetation and birds as homogenizing communities within cities due to human management and disturbance (McKinney 2006, Aronson et al. 2014), cultivated soils have been overlooked as a strong ground up force that select for vegetation systems and associated biodiversity to develop. Soils, not only vegetation, are a foundational catalyst for the emergence of novel community garden agro-ecosystems.

Community gardens as social-ecological systems

Community gardens serve both social and ecological goals of the city thus adding to urban landscape multifunctionality (Lovell and Taylor 2013). On one hand, community gardens provide a place where urban dwellers can interact with nature, increase food security, and contribute to public/human health and social well-being; on the other hand, they have an important role in combatting the increasing imperviousness of the built environment, providing habitat heterogeneity in the landscape matrix, and supporting urban biodiversity. In the following section, we describe the key social and ecological characteristics of community gardens and how they interact to provide social-ecological benefits.

Community gardens as social systems

Although historically practiced by ancient civilizations, urban agriculture has more recently “re-sprouted” in developed and developing cities, transforming grey and green land into highly productive agro-ecosystems that feed urban residents and add multifunctional complexity to urban landscapes (Lovell 2010). In the past, Mayan civilizations integrated urban agriculture into urban planning and design to contribute to food security, sustainable water management, and urban welfare (Barthel and Isendahl 2013). Today, urban agriculture is in its “fifth wave” in the Global North, and is promoted by alternative agriculture networks and local community movements as a mean to increase urban food security, literacy, and justice (Lawson 2005). In particular, community gardens have been advertised as a critical solution to decrease social inequities and increase green space access in urban landscapes. As a result, contemporary community gardens are now providing public health (Alaimo et al. 2008), cultural and social benefits (Saldivar-Tanaka and Krasny 2004) traditionally linked to food production. Some benefits may not have been originally present historically (e.g., multicultural interactions), whereas others may have been lost in modern history (e.g., people-food-nature connections).

Gardens generally build community capital through increased social cohesion, reciprocal support, and interpersonal connections. While urban societies are increasingly socially disconnected due to people isolation and limited time spent outdoors (Mckenzie 2008), agro-ecological participation offers gardeners the physical and social space to develop a sense of place, build social networks, and thereby develop relations of reciprocity and trust (Kingsley and Townsend 2006). Increased social interactions foster community cohesion and inclusion around a shared activity (Alaimo et al. 2010). Furthermore, gardens are locations of neighborhood community organizing and political activity (Armstrong 2000), and gender political empowerment (White 2011). This suggests that garden social network activity can have spillover effects on the overall urban neighborhood welfare. Thus, gardens may foster a novel community connectedness that contributes to mending the rifts generated by capitalist economies of scale and urbanization (McClintock 2010).

Social connectedness has been shown to enhance agricultural engagement and learning across cultures and generations that translate into better gardener management practices (Saldivar-Tanaka and Krasny 2004). Gardeners not only share physical objects like gardening tools and fresh produce (Wakefield et al. 2007), but also share their knowledge capital rooted in past rural livelihoods (Baker 2004). Gardeners exchange ideas and solutions to overcome challenges like soil contamination (Kim et al. 2014), poor soil fertility, and the lack of access to compost or fertilizer (Baker 2004). The transfer and adaptation of knowledge on soils and vegetation within gardening communities may generate resource conservation and food production. Thus, community gardens are sites of knowledge gathering, education, and collective knowledge co-production – all of which are novel processes in diverse (in cultures, ages, and socio-economic status) gardener communities.

Perhaps less well known are the health and mental well-being benefits that may come from community gardens. Research has shown that the deprivation of urban nature has lead urban citizens to experience: 1) psychological suffering in the form of greater stress and anxiety (Lederbogen et al. 2011); and 2) physical health effects in the form of increased allergies, depleted human microbiota and decreased immunoregulation (Hanski et al. 2012). Community gardens may decrease stress and provide improved mental well-being and cognition through an increased interaction with urban nature (Van Den Berg and Custers 2011). In addition, community garden participation may improve the physical health of gardeners through exposure to highly biodiverse systems or through the cultivation and ingestion of medicinal plants that combat sickness (Corlett et al. 2003). Gardening can also improve the nutrition of participants, especially of children, by increasing the amount and diversity of vitamins, minerals, and phytochemicals gained through increased fruit and vegetable consumption (Guitart et al. 2014). Last, gardeners' exposure to natural elements in urban environments may change their perception of biodiversity. Experiencing nature further improves human well-being, happiness, and can have implications for biodiversity conservation (Russell et al. 2013). In gardens, novel interactions with garden biodiversity (e.g., plants, insects, soil-dwelling organisms) in both a recreational and productive manner may enhance an individual's perceived connectedness to nature (Dallimer et al. 2012) and influence their agricultural management (Andersson et al. 2007).

Community gardens as ecological systems

Community gardens are increasingly recognized as reservoirs for urban biodiversity conservation (Goddard et al. 2013) and as laboratories for ecological investigations that explore the influence of management practices on biodiversity, trophic interactions, and ecosystem service provisioning (Philpott and Bichier 2017). Local vegetation and soil management influence above- and below-ground habitat characteristics, ecological interactions and processes through soil amendments, irrigation practices, and the intentional planting of resource plants (Fiedler et al. 2008). Additions of floral and ornamental plants provides nectar and resources to support beneficial insect populations and species diversity in gardens (Colding et al. 2006), which enhance crop production through, for example, pollination and pest control (Lin et al. 2015). Soil management (e.g., adding compost, fertilizer, manures) and groundcover management (e.g., mulching) increase soil decomposition rates, fertility, organic matter development, moisture and water retention (Beniston and Lal 2012). Together, soil and vegetation management determine the template for ecological community formation and trophic interactions (Faeth et al. 2011).

Ecological interactions among and between trophic levels lead to above-ground (e.g., pollination) and below-ground (e.g., stormwater infiltration) ecosystem services that encapsulate supporting, regulating, cultural and provisioning services. Garden soils are hotspots of nutrient cycling and the development of soil fertility. Natural enemies respond to local garden management practices as well as landscape patterns, which are good proxies of garden pest population regulation (Philpott and

Bichier 2017). The control of herbivorous and rodent pests is provided by insectivorous and predatory birds that may use community gardens for nesting habitat and food resources (Goddard et al. 2010). Most often cited are the provisioning services of food, water, and biodiversity generated through ecological functions linked to the management of soil and vegetation that support e.g., pollinator communities that enhance crop production. Last, in addition to conserving biodiversity and biotic functions, community gardens are also important for resource conservation, including water storage through vegetation and soil structure (Davies et al. 2009). Gardens may also regulate within garden microclimate and surrounding neighborhood climate to lessen urban heat island effects that result from impervious cover (Drescher et al. 2006). In sum, all of these ecosystem services have novel social-ecological spillover benefits to the surrounding urban landscape (Lin et al. 2015).

Novelty created through the cultivation of garden soils

Urban soils tend to be very different from “natural” and agricultural soils resulting from the biophysical effects of urbanization like compaction, chemical pollution, and heat and precipitation retention (Byrne 2007). However, gardens present another layer of complexity as they are formed out of social, political and economic dimensions of the human community (WinklerPrins and Souza 2005). This suggests that community garden soils are highly influenced by diverse management practices, socio-economic gradients, and biophysical conditions. Soils thus provide not only a link between above- and below-ground trophic interactions, but are also the layer where gardener socio-cultural diversity and social and economic capital fuse together to influence soil formation. Consequently, soils are regulated not only by soil food web interactions, but also by garden social networks and individual resource availability (e.g., tools, time, money) (Ghose and Pettygrove 2014). While natural soils are typically characterized through their soil organic matter, moisture, and nutrient content, garden soils are novel in that they must also be understood in the context of the gardener's knowledge capital, fiscal capital, and social capital that influence soil management.

Community gardens and their soils may further be influenced by historical legacies of industrial or agricultural land-use (Palmer and Santo 2016). Community gardens located on previous industrial sites often have soils that contain heavy metals, have low organic matter content, and have high levels of compaction (Pouyat et al. 2010). Consequently, community gardens often need to be re-created *from the ground up* in order to make them appropriate for cultivation. It is this social and biophysical process of soil creation that forms the ecosystem base in which vegetation and other species thrive. In sum, garden soils sit at the interface for human and non-human interactions through micro-managed soil development. Soils sequester a novel materiality and a social production of ecological systems through this process that render them unique from other novel ecosystem accounts.

A useful concept to contextualize these local and landscape scale processes is *anthropedogenesis,* the role of human activity in the process of soil formation

(Richter 2007), that is rapidly occurring in urban landscapes (Effland and Pouyat 1997). Previous work has demonstrated how social context, history, and management regimes in which urban soils are embedded influence soil properties across urbanization gradients (Effland and Pouyat 1997, Swidler 2009). For example, urban soil contamination may be a product of landscape scale legacy effects of uneven urbanization processes, linking historical waves of capitalism to contemporary racialized environmental injustices (McClintock 2015). In addition, soil transportation may connect cities to their rural hinterlands to create novel connections across space and time. For example, rapid land-use transformation (e.g., housing abandonment and demolitions) and material constraints (i.e., for filling material) has demanded the extraction and movement of fresh soils from rural areas to fill demolished building lots in Detroit, Michigan (Koscielniak 2016). These pedogenesis stories highlight how soils, novel in composition and spatial association, form from interactions between urban ecosystems and urban society.

Previous work has focused on the role of vegetation as a primary mechanism for influencing novelty in urban environments (Kowarik 2011, Aronson et al. 2014). We instead consider the effect of garden soils as the primary mechanism of novelty within community gardens. In gardens, urban biophysical and socio-economic activities that influence soil development set filters for species selection to shape biodiversity and ecological processes through the provision of water and nutrients. We see the potential cascading effects on vegetation complexity, associated biodiversity, and above-ground trophic interactions that provision ecosystem services through community garden soil cultivation as a distinct account of novel ecosystem development in comparison to natural and agricultural landscapes. Below we provide a case study example to show how these patterns have emerged within the community garden soils in central California.

Community garden soils as emergent phenomena: the case of California's central coast

The California central coast landscape is a mix of intensive agriculture, high density urban development, and forested mountain ranges that create sharp physical and climatic distinctions within and across cities. The biophysical and climatic gradient is mirrored by a socio-economic gradient. To the south, the stretch of landscape from Santa Cruz to Watsonville exemplifies the irony of California's agricultural system; thousands of acres of strawberry and lettuce fields are cared for by farm workers, many who live in communities with extremely limited access to fresh produce (Brown and Getz 2011). To the north, the influx of Silicon Valley wealth has fundamentally changed the physical and social structure of a historically orchard-dominated landscape tended by Asian and European immigrants (Pellow and Park 2002). These historical waves of land-use transformation and demographic transformation shape the contemporary diverse physical and social composition of central coast cities.

We present a case study example based on 25 urban community gardens across the central coast of California (Figure 1) that experience a large range of sociodemographic, cultural, and ecological heterogeneity. Within each of the gardens, gardeners rent individual garden beds that they manage based on their own desires (e.g., crop choice, ornamental plantings, levels of amendments), but also under the prescribed management of the overall garden. For example, some gardens have water limits, while others have no water use restrictions. Some gardens provide compost whereas others provide mulch. However, we found that all gardens are managed towards the cultivation of organic produce, where garden management (e.g., city parks and rec departments) only allows the use of organic inputs provided by the garden or brought in by the individual.

The heterogeneity in social and environmental factors is expected to yield a range of management decisions that affect soil composition and vegetation choices across gardens. Some gardens are primarily cropped with edible plants, whereas others are planted with ornamental species. Some gardens have drip irrigation systems, while others use watering cans to transport water from a main spigot or garden hoses from shared spigots distributed across the garden. Some gardeners choose to use raised beds or boxes, while others garden in the ground, and gardeners choose how to compose the soil within their beds. Gardeners choose the base soil material as well as the amendments added to beds (compost, manure, mulch). Yet, gardeners may or may not add amendments to their beds based on resource access or perceived necessity, and the quality and composition of amendments likely varies as a function of the social context and market availability.

Thus, because garden beds are managed individually by gardeners, we have chosen the garden bed as the scale of analysis in which to contextualize how land-use history, chronology, and potentially the management drivers (e.g., knowledge sharing, fiscal capital) and environmental drivers (e.g., regional climate, parent material, soil structure) of the gardens described above extend to garden soil characteristics. We see these social complexities situated in the context of urban landscape biophysical heterogeneity as the driver of soil development in community gardens (Figure 2). Further, the diversity of gardeners, their motivations and values, and their social and economic capital contribute to the production of soils that support a novel agro-ecosystem within the urban sphere.

Remediating history: disappearing land-use legacies

Our research in California has unveiled that contemporary community gardeners can mediate the effects of land-use histories and chronologies of urban soil. Triplicate samples (20 cm depth) of the topsoil (re)worked by community gardeners were used to measure physical and chemical soil properties that affect plant growth and crop yield. Total C and N were measured with a CE NC2500 Elemental Analyser interfaced to a ThermoFinningan Delta Plus XP isotope ratio mass spectrometer (Fry et al. 1992) at the UCSC Stable Isotope Laboratory (Santa Cruz, CA, USA). Soil extractable Fe, Zn, Cu, Mn, B were measured using DTPA – Sorbitol Inductively Coupled Plasma Spectrometry (Gavlak et al. 2003) at A&L Western Agricultural Laboratories (Modesto, CA, USA). Soil organic matter (SOM), maximum water holding capacity (WHC) and bulk density (BD) were measured using loss-on-ignition (550°C, 3 h), water bath and core methods, respectively (Wilke, 2005). We also collected information on previous land-use (PLU) of garden sites using aerial imagery dating back to the 1940s (www.historicaerials.com). We determined the PLU (agriculture/pasture, forest, housing, tree orchard, and vacant lot/open space), the minimum year of PLU, decade of garden establishment, and garden age. Despite predicting that garden soils would differ as a function of PLU type and age due to legacy effects, we did not find significant differences among sites of different PLU type or age group (Figure 3). This indicates that legacy effects typical of urban soil characteristics (Raciti et al. 2011), such as those observed in turfed systems (Pouyat et al. 2009), may be lost due to the high level of cultivation, manipulation, and management by gardeners. We further found that cultivation by community gardeners might generate relatively novel ecosystems in raised garden beds as compared to those established onto the ground surface, the former having generally higher soil organic matter content, carbon and nitrogen (Figure 3).

The results indicate that the land-use legacy traits are not maintained within the soil systems, and that garden age does not have a large impact on garden soil signatures. This indicates that management or creation of soils within community gardens is homogenizing the soils to a point where past soil characteristics are progressively lost. Garden soils are thus novel in that they are disassociated from past land-use legacies of that geographic location. However, soils from raised beds compared to ground beds are significantly different from one another, indicating that different socio-ecological processes might differentiate the two systems.

Furthermore, the cultivation and development of "garden" soil – especially in the case of the development of raised beds – are increasing soil homogenization and novelty. Observations from field work indicated that gardeners often bring base soils from other locations, such as large home gardening stores or council supplied soil collections. The construction of raised beds further provides a gardening system where soils can be completely cultivated outside of the original soil profile, thus allowing gardeners to create novel soils exactly to their liking. As we see, the soils in the raised beds have higher soil organic matter, carbon, and nitrogen. The lack of strong differences in raised bed soil characteristics across regions – each of which has different microclimate characteristics, socio-demographic gradient, and degree of urbanization (i.e., percent impervious cover) – also further substantiates that these highly cultivated and novel soils are quite homogenous in quality and structure. This suggests that people's practices, knowledge, and attitudes when it comes to soil cultivation are relatively similar.

Previous research has found historical land-use transformations drive urban soil characteristics. For example, legacies of industrialization have been linked to soil mercury contaminants in urban Oakland, CA (McClintock 2015). Soil nitrogen and carbon densities have been found to increase in urban developed soils in comparison to natural forest soils (Raciti et al. 2011). Further, temporal factors have been attributed to anthropogenic drivers where, for example, organic matter increases with urban soil age (Scharenbroch et al. 2005). The lack of strong temporal relationships in our system parallels other studies of heavily managed urban green spaces. Golubiewski (2006) found urban soil carbon progressively increases (actually exceeding surrounding native soils) and bulk density decreases after decades since land-use change. Livesley et al. (2016) found that turfgrass and urban forest soil properties were better predicted by local vegetation factors than green space age. These examples indicate that local management of urban soils may be more important than age by ameliorating negative effects of urbanization like soil compaction (Millward et al. 2011).

In our California system, the high cultivation of garden soils seems to resolve the influence of past land-use legacies in a relatively short time period (<15 years). High management intensities at different spatial and temporal scales seem to converge the disparate trajectory of original soils into novel systems. While we observed relatively high soil compaction in sites that have histories of mechanized agriculture, the ability of community gardeners to work and re-work their soil, utilize raised beds and add mulch and organic amendments may negate strong universal effects of land-use transformations on garden soil characteristics and actually lead to the homogenization of soils across sites and regions. Furthermore, in comparison to soils in other urban green spaces like parks and recreational areas, this increased management intensity in the case of gardens may actually be advantageous rather than a force with negative connotations (Edmondson et al. 2014). Gardeners may rapidly ameliorate poor soil quality, and increase its fertility and productivity in urban areas otherwise unfavourable to food production.

Conclusion

Garden soils, due to their high level of cultivation and management, determine novel agro-ecosystems characterized by a unique soil structure and quality desired by gardeners. Community gardeners, thus, become agents of change establishing the primary material within ecosystems that mediates below- and above-ground interactions. Soil characteristics are a function of specific management practices, such as adding mulch cover and compost, and these management practices are likely a result of gardener social and physical capital linked to social identity, socio-economic status, and social networks themselves novel in cityscapes. We suggest that the practices and knowledge of soil cultivation – and in particular the formation of raised beds – are similar across gardener communities to thereby drive soil homogenization and the loss of land-use transformation legacies and historical contingencies. Instead, novel garden soils provide the basis for continuous land and habitat transformation with the high levels of manipulation, cultivation, and remediation by gardeners.

Figure 10-1. California research sites located within 25 community gardens (green markers) in Santa Clara (San Jose), Santa Cruz, and Monterey counties. The landscape has experienced rapid land-use transformation over the course of the last century – from natural, to agricultural, to urban land-use. One garden, Berryessa Community Garden, was once dominated by orchards and agricultural fields. Now it is a residential neighborhood. Images courtesy: Google Earth™ and Historic Aerials.

Figure 10-2. Community gardens are constructed using different soil inputs and amendments (a, b), are managed using different groundcover and water conservation techniques (c, d), and are often housed in raised beds (e-g). Photos courtesy: M. Egerer.

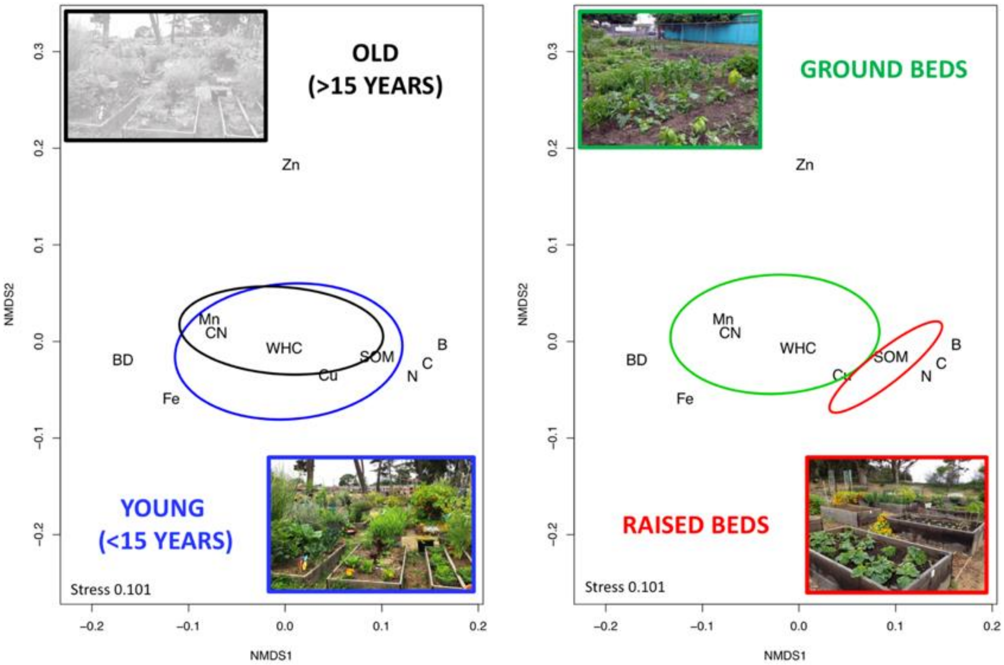

Figure 10-3. Ordination of the California central coast community gardens based on Bray-Curtis distance among their soil properties (bulk density (BD), total carbon (C), total nitrogen (N), C:N ratio (CN), total zinc (Zn), total manganese (Mn), total iron (Fe), total copper (Cu), total boron (B), soil organic matter (SOM), and water holding capacity (WHC)) using non-metric multidimensional scaling. Community gardens are classified by age (left panel) and garden bed type (right panel). Plots were created using the "ordiplot" and "ordiellipse" functions of the R library *vegan* (Oksanen 2015) by adding 95% confidence ellipses around each age and plot structure group.

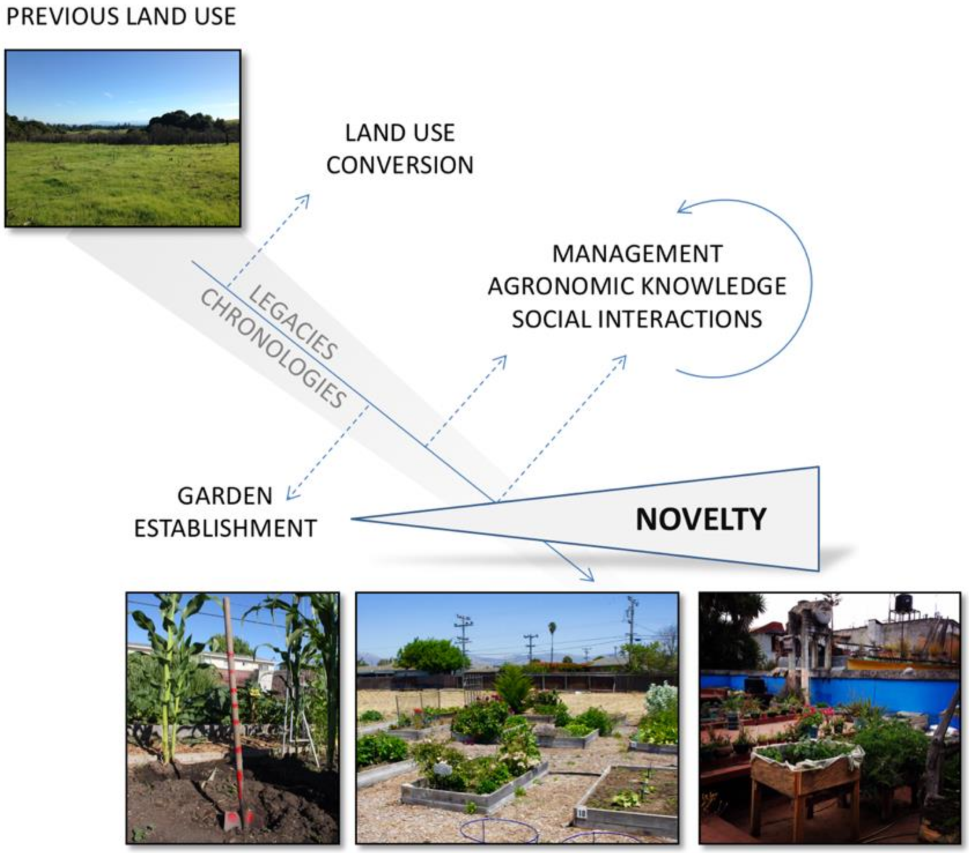

Figure 10-4. The creation of socio-ecological novelty in community gardens is a concept transversal to time and management. Management, knowledge and social interactions change through time (linear arrows) but also affect each other (circular arrow). The three pictures symbolize ground, raised and “off the ground” beds as an example of novelty and departure from the original background (or historic) conditions. Of note, while “off the ground” systems were not investigated in our California study, we see this as a component of the novelty continuum.

Conclusion

Urban agroecosystems are poorly understood because of their social and ecological complexity. The interplay among local agricultural management and landscape features influence the ability of urban habitats to support biodiversity and provide ecosystem services (Colding et al. 2006). However we know little about how land use management relates to urban social dimensions in urban agroecosystems (Goddard et al. 2010). Few have studied the ecology in urban gardens and the sociology of urban gardens in parallel at both scales (Gregory et al. 2015). This has left gaps in our understanding of the relationships among human diversity, management, and agrobiodiversity that provision ecosystem services and contribute to human well-being (Guitart et al. 2012). Only recently have scholars begun to use agroecology as a framework to understand urban agroecosystems (Altieri and Nicholls 2019), but in limited contexts (Hermann et al. 2018). These efforts are just a start, and we still require paralleled agroecological knowledge of rural systems in urban systems in order to improve sustainable agricultural management in the city.

The collective work herein aims to contribute to filling this knowledge gap. I use methods from ecology and conservation biology, agroecology and geography to understand the specific local and landscape scale drivers of biodiversity, ecosystem services and urban sustainability in urban agroecosystems: community gardens. Here, ecology and conservation biology provide the theories and hypotheses to predict patterns of species diversity and distribution, and species interactions and ecological functions in response to local and landscape environmental drivers. Agroecology provides not only the foundational agricultural context, but the framework necessary to explicitly relate ecological theory and sustainable agricultural production to one another and to human well-being. Geography provides the means through which to assess how landscapes and the people within them shape one another through political, economic and environmental processes.

Relatively few scholars have provided interdisciplinary syntheses that blend these fields of knowledge. Indeed, my own synthesis of the ecology, agroecology and geography of urban gardens now comes in this conclusion of several studies. Though some may find that this work lacks focus, I argue that this is the crux of doing interdisciplinary scholarship in an academic structure designed by disciplined scholars. Moreover, scholarship is situated in a world in which we increasingly understand phenomena and the world around us in bite size pieces, in short stories, and in 200 characters. Telling a scientific story isolated from certain events, factors, data, can focus the narrative, prevent distraction, and simplify the outcomes. But synthetic conclusions take more effort to piece together the whole story, to unpack all of the complexity and then try to explain the logic behind the order and structure. I now aim to summarize, explain and synthesize the findings from this work, and end with implications.

The novelty of urban landscapes and the ecological communities and interactions within them may or may not follow theories in ecology that predict, for example, species-area relationships and the dynamics between predator and prey (Kowarik et al. 2011). Landscape scale habitat fragmentation, certain environmental factors, and human management decisions drive land cover (vegetation) composition at local and landscape scales (Cadenasso et al. 2007; Williams et al. 2008). Socioecological filters are therefore also important to describe patterns of biodiversity and ecosystem functions (Aronson et al. 2016) because they may drive ecosystem convergence and biotic homogenization to reduce biodiversity (Groffman et al. 2014) or weaken the effects of historical land use legacies (chapter 9). Moreover, climatic and biogeochemical characteristics of urban regions show novel fluxes and patterns. Urban heat island effects and supplemental nutrients and irrigation changes the seasonal distribution and availability of resources (Grimm 2008) to affect patterns of biodiversity (Parris and Hazell 2005; Aronson et al. 2016). Moreover, the pollution of soils, air and darkness can influence the biodiversity within habitats and organisms' habitat use by altering their activity or filtering certain traits (Shochat et al. 2010; Faeth et al 2012). Therefore, both environmental (abiotic, biotic) and social factors at local and landscape scales drive community assembly processes in the city (Williams et al. 2009; Aronson et al. 2016). Because biodiversity (species diversity, community composition) generally relates to the ecosystem function of the system and subsequently how people use and benefit from an urban habitat, hierarchical filtering of species through socioecological forces will affect ecosystem service delivery and human well-being.

Work has shown that the local and the landscape factors affecting biodiversity are context dependent on one another (chapter 1). I provide another good example of local-landscape relationships using ladybird beetles as a model system. In chapter 2, I show that local management factors are important predictors relative to the context of the landscape surroundings. In this system, the amount of impervious surface surrounding the garden – describing how "urban" or built the landscape is – is driving species abundance, richness and in turn the relative importance of other local management factors (e.g. groundcover, crop diversity, tree and floral availability) for lady beetles. Moreover, impervious cover is significantly influencing the fidelity of lady beetles to gardens. Few studies have found higher species richness in urban agroecosystems in relation to natural areas (Rees et al. 2009; Philpott et al. 2014; Burkman and Gardiner 2014; Hall et al. 2016), and increases in species diversity with urban disturbance (Helden and Leather 2004). These studies suggest that urban agro-/ecosystems provide particular limited resources that are less available in the urban matrix and that urban habitats can be critical for biodiversity conservation. In this dissertation, I show that ladybeetles are more abundant and diverse in gardens in a sea of impervious surface, but have lower fidelity to those same gardens. Ladybeetles are not dispersal limited by the urban matrix but are spatially independent of garden habitats. Thus what particular limited resource(s) do gardens have that is less available in natural areas that may drive their affinity to gardens, but not necessarily their fidelity to gardens?

I conclude that ladybeetles are highly mobile organisms that are spatially independent of habitats. Ladybeetles are also likely utilizing resources within the urban matrix to an extent. Given this promiscuity, gardens do have something that is rather abundant in combination with other habitat requirements in comparison to natural areas and other urban greenspaces: supplemental water. Indeed, when we compare our findings to ladybeetle observations in a system with similar urbanization patterns but an inverse precipitation pattern, we find the very opposite result: beetles are fewer and less diverse in gardens in more urban landscapes. Here beetles may not overcome the novel challenges that urban landscapes present arthropods as they can survive elsewhere. To consider other life history characteristics, while in winter Michigan beetles retreat to heated residential homes to evade the cold, in California many beetles retreat to the rainy mountains to congregate, drink and mate (Gordon 1985). Thus in both regions, anthropogenic factors (altered irrigation, heat) may drive inverse related patterns of ladybeetle diversity and distribution by providing a limited resource within that specific environmental context.

If water availability can drive patterns of biodiversity, biodiversity that is important for ecosystem service provision, what drives the irrigation of these habitat oases in the otherwise arid matrix? What local and landscape factors are important for water use and water conservation? Global environmental changes in climate and land use are challenging the environmental and social sustainability of urban gardening – which relies heavily on water availability – in these regions. Future projections of increasing climate extremes (e.g. drought, heat) is making water use and conservation a local, regional and global issue (Vorosmarty 2000). Particularly in the western U.S. (Seager et al. 2007) and areas of similar climatic regimes. In urban regions, city policies and planning are incorporating water conservation strategies and infrastructure into future sustainability plants (e.g. Phoenix, AZ) but are also mandating water use restrictions during extremes (e.g. San Jose Parks and Recreation 2016). Yet these policies generally conflict with the empirical evidence that residents are more likely to use more water during those climatic periods (Balling and Grober 2007; Balling et al. 2007), particularly for garden irrigation (Lin et al. 2018).

Chapters 6, 7 and 9 add to this conflicted evidence. I show that gardens are very irrigated habitats in times of drought, and gardeners have little understanding of how much water that they use (chapter 6). Moreover, gardeners are more likely to use more water in the likelihood of more frequent and intense heat and drought events (chapter 7); gardeners agree that watering behaviour changes with extremes, but less so on changing their planting practices. Irrigation practices have social and ecological consequences. Irrigation may attract and support biodiversity that require water to live and reproduce in the agroecosystem. Irrigation is of course important for plant production, although this may also be linked to higher herbivore pest abundance (chapter 4). However, irrigation can cause problems in sustainable water governance within resource stressed communities: rules may indeed lower average water usage by gardeners (chapter 7), but water governance within gardens of diverse membership can result in racialized conflicts that challenge the idealized notion of community gardening as an egalitarian endeavour (chapter 8), present in the urban agriculture/agroecology social movement discourse. Thus what is free water for ladybeetles to imbibe upon, is not free from the capitalist logics that underpin resource flows in urban landscapes and every day experiences of and conflicts among urban people (Kaika 2003). It is important to trace the drivers of resource use and their influence on social and ecological processes to understand potential synergies and trade-offs. I argue that water – in its material form, in its use – exemplifies the synergies and trade-offs among biodiversity (predators, plants and pests), ecosystem services (water conservation, food production, pest control), and human well-being (social cohesion, basic materials to live a good life, freedom of choice and action) (MEA 2005). Future work can directly assess ecosystem service relationships among facets of the agroecosystem that influence their sustainability.

To conclude, how do we grow food in the city in a form that optimizes synergies among biodiversity, ecosystem service and human well-being relationships that is environmentally and socially sustainable? What are the collective lessons learned from these case studies within urban agroecosystems? How may the lessons inform an urban agroecology? Gardens are unique from other green spaces (e.g. parks) in the ecosystem services that they provide, in that they provision food. Moreover, most urban community gardens are explicitly designed for organic food production, requiring context dependent sustainable practices. Yet, garden governance and management may ironically agree with neoliberal logics of urban as well as agricultural landscapes (chapter 7). The complicated socioecological lives of gardens—the membership rules, water rationing, and theft of produce—illustrate the challenges and the agonisms embedded within garden communities. Nevertheless, untangling these agonisms can also reveal how gardens can be places where people bring together ideas on how to answer the social and environmental problems facing urban environments.

Some of these answers lie in the practices, the consciousness, and the self-awareness of the practitioners (urban gardeners). First, practitioners should acknowledge the important role that they play as stewards of urban biodiversity conservation, users of resources, and as representatives of a greater social movement (whether they wish to be or not). Second, practitioners could collectively discuss as a community what they recognize as ecosystem services to them, and evaluate how to practically manage for synergies and trade-offs as perceived by them, given the knowledge of these relationships. For example, gardeners can negotiate whether to design garden landscapes to optimize wild bee pollination (e.g. leaving bare soil for ground nesters) or more towards water conservation (e.g. instituting watering rules; providing straw resources) or for social cohesion and knowledge exchange among participants (e.g. having spaces to rest or play). While some gardens may be able to enhance multiple and diverse services, others may have to decide on trade-offs.

Other answers lie in urban planning and policy. Policies must value and protect urban agroecosystems for their innate ecological/biological value (biodiversity conservation), and their benefits to human society through ecosystem services and improvements to human well-being. Demonstrating that gardens in highly urban landscapes are especially important for conserving certain organisms as well as for food production and well-being benefits provide evidence for this argument and reasonable justification. Incorporating more and diverse green infrastructure within – rather than at the periphery of – urban landscapes is essential to increase the flows of these said benefits across the landscape and across social gradients for just service provisioning.

Last, answers lie in the scientists who work in the intersection between social-ecological theory, applied knowledge production, and social-environmental change. Ecologists and interdisciplinary scientists play a fundamental role in advancing agroecological knowledge to cultivate sustainable urban food systems. Moreover, scientists can work hand-in-hand with practitioners and planners to co-generate applied outcomes for sustainable urban landscapes, not only ecologically interesting outcomes for “science” (Tanner et al. 2014).

Promoting change at these three scales will have on-the-ground positive effects on sustainable urban agricultural management, on urban welfare, and on furthering urban agroecology as a science, practice and social movement.

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Chapter 10

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