Title
The Biodiversity and Direct Ecosystem Services and Disservices of Urban Gardens

Author
Clarke, Lorraine Weller

Publication Date
2014

Peer reviewed|Thesis/dissertation

UNIVERSITY OF CALIFORNIA
RIVERSIDE

The Biodiversity and Direct Ecosystem Services and Disservices of Urban Gardens

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

Doctor of Philosophy

in

Plant Biology

by

Lorraine Weller Clarke

June 2014

Dissertation Committee:
Dr. G. Darrel Jenerette, Chairperson
Dr. Edith Allen
Dr. Derick Fay

The Dissertation of Lorraine Weller Clarke is approved:

Committee Chairperson

University of California, Riverside

Acknowledgements

First of all, I would like to thank my advisor, Darrel Jenerette, for always pushing me to be my best and do great science. He has been supportive, constructively critical, and encouraging in my journey during these past 6 years. In addition, I would like to thank the members of my dissertation committee, Edith Allen and Derick Fay, for their support and feedback on my work and willingness to participate in the research process. I also thank the hard-working administrators and office staff in the Botany and Plant Sciences program, especially Deidra Kornfield, who have helped me through many technical issues. I have had the great opportunity to work with several great collaborators outside of my institution and have written papers and done field research with many of them. These include Liangtao Li and Zhenrong Yu at China Agricultural University, Daniel Bain at the University of Pittsburgh, Diane Pataki and Meghan Avolio at the University of Utah, and Stephanie Pincetl and Tom Gillespie at UCLA. For field research and data support at the multiple institutions I worked at, I thank Cara Fertitta, Lauren Velasco, Jennifer Eberwein, Lindy Allsman, and undergraduate members of the Jenerette lab, and Li Xiang and members of the Yu lab in China. For facility support, I thank the University of California, Riverside, China Agricultural University, and the University of Pittsburgh. I also thank Norm Ellstrand, and Exequiel Ezcurra for ongoing research discussion. Finally, I thank UC Riverside herbarium director, Andrew Sanders for extensive aid in species identification and archiving samples.

The projects in this dissertation were supported by the US National Science Foundation (DEB 0919006, OISE-1210137, DEB-1210953), the Chinese National Research Project (2012BAJ24B05), UC Mexus, China Agricultural University, and the University of California, Riverside. The text in Chapter 3 of this dissertation is a reprint, in part, of the material as it appears in Urban Ecosystems, March 2014 (Clarke, L.W.; Li, L.; Jenerette, G.D.; Yu, Z. 2014. Drivers of plant biodiversity and ecosystem service production in home gardens across the Beijing Municipality of China. Urban Ecosystems DOI 10.1007/s11252-014-0351-6). The co-author, Liangtao Li, listed in that publication organized and aided in the field research. G. Darrel Jenerette directed and supervised the analysis and writing of the paper in the U.S., while Zhenrong Yu organized and directed data collection which occurred in Beijing, China.

Dedication

This dissertation is dedicated to my loving husband, David Clarke. His constant love, support, and encouragement have helped my research thrive.

ABSTRACT OF THE DISSERTATION

The Biodiversity and Direct Ecosystem Services and Disservices of Urban Gardens

by

Lorraine Weller Clarke

Doctor of Philosophy, Graduate Program in Plant Biology
University of California, Riverside, June 2014
Dr. G. Darrel Jenerette, Chairperson

Urban agricultural systems, like community and home gardens, may act as oases of biodiversity in cities dominated by impervious surfaces. They have also been shown to bridge gaps in food security and provide socio-cultural benefits. Despite these benefits, little research has been conducted that evaluates factors influencing garden plant biodiversity and ecosystem services (ES). Also less intensively researched are ecosystem disservices that gardens can contribute to, like gardener exposure to heavy metals. Urban soil can act as a sink for heavy metal contamination, which is mostly deposited through anthropogenic pollution. This dissertation addresses knowledge gaps about ES with two comprehensive surveys of garden biodiversity and ES production, one on community gardens in Los Angeles (LA), CA and one on home gardens across an urbanizing gradient in Beijing, China. It also addresses disservices and bioavailability of three heavy metals (lead, arsenic, and cadmium) through a soil survey and sequential analysis of heavy metals in LA gardens.

My main results indicate an overall shift in biodiversity from provisioning (food and medicinal production) to cultural (ornamental) services with increased gardener income and access to city resources (like grocery stores or markets) in both U.S. and

Chinese gardens. This result supports a hierarchy of need, where gardeners preferentially plant species that support their most pressing needs, like food security. Urbanized regions in Beijing and immigrant-run gardens in Los Angeles also formed culturally distinct assemblages of edible species based on shared agricultural experiences. As the most common use for species was food, understanding metal bioavailability is important for accurate risk assessments. Lead, particularly in reducible form, increased the most with age of neighborhood, indicating oxidized lead paint buildup. Cd and As exchangeable fractions increased with proximity to road, indicating sources from air pollution. Finally, while Cd became less bioavailable with increased organic matter, reaction with organic humic acids released reducible As into the bioavailable fraction. These quantitative results can inform land managers about valued biodiversity and provisioning service from gardens in food insecure regions, as well as valuable information on how to predict metal accumulation hotspots and reduce plant uptake of metals.

Table of Contents

List of Figures

List of Tables

Chapter 1: Introduction

Urban gardening has been integral to city life throughout the world since prehistory (Hynes 1996, Fedick 1996, Stark and Ossa 2007). One prominent type of urban gardening is the community garden, defined as urban land gardened by multiple residents (Colding and Folke 2006). Recent surveys suggest 10,000 community gardens are functioning throughout the U.S. with more than 1 million U.S. participants (Lawson and Drake 2013). Another common urban agricultural system is home gardens, complex, multi-layer systems of trees, shrubs, and crops around homesteads (Kumar and Nair 2004; Michon and Mary 1994; Del Angel-Pérez and Mendoza 2004). Globally, agricultural regions within large cities range between 16% (Stockholm, Sweden; Colding et al 2003) and 36% (Dunedin, New Zealand: Mathieu et al. 2007) of total land area. Urban agricultural systems may act as oases of biodiversity in cities dominated by impervious surfaces (Colding et al. 2006; Gaston and Gaston 2011). They have been shown to bridge gaps in food security and nutrition (Alaimo et al. 2008), benefit soil nutrients and cycling (Zhu et al. 2006), and provide socio-cultural benefits (Kingsley et al. 2009). These ecosystems are increasingly the focus of coupled human-natural systems research (Kirkpatrick et al. 2007; Aguílar-Støen et al. 2009; Bernholt et al. 2009; Kabir and Webb 2009) with increased scientific demand for quantification of garden plant species abundance, community diversity, ecosystem functioning, and ecosystem services (Guitart et al. 2012; Huai and Hamilton 2009; Jaganmohan et al. 2012). One major risk factor for garden participants is the prolific availability of heavy metals in urban soils

(Mielke et al. 2010). Metal availability can impact both food production and safety of food grown in these plots, as well as the long-term viability of urban soils.

Though research on community and home gardens has been increasing, many studies are qualitative and descriptive (Draper and Freedman 2010, Guitart et al. 2012), and few studies actually address the effect of urbanization and income on production of direct ecosystem services, like food or medicines (Guitart et al. 2012, Jaganmohan et al. 2012). The interaction of human input and heavy metal contamination in urban environments also remains undercharacterized, with more published research on exposure rates, not mechanisms influencing metal bioavailability and contamination sources (Charlesworth et al. 2010; Dianoco and Montemurro 2011; see Yesilonis et al. 2008 and Schwarz et al. 2012 for exceptions). My presented dissertation research is interdisciplinary, incorporating ecology, anthropology, and sociology to address these knowledge gaps. I focus on quantifying biodiversity and ecosystem services produced by community and home gardens, specifically the direct services of food production and aesthetic value and disservices of urban heavy metal dynamics.

Ecosystem Services

My dissertation focuses on the production of ecosystem services and disservices in urban garden ecosystems. Humans are reliant on ecological processes to provide food, water, climate mediation, and even cultural and aesthetic value; these benefits have been termed "ecosystem services" (Millennium Ecosystem Assessment, [MEA] 2005). These can be subdivided into provisioning (food, water, timber), regulating (those affecting disease, climate, flooding), cultural (recreational, aesthetic, spiritual) and supporting (nutrient cycling, photosynthesis) services (MEA 2005). Ecosystem disservices occur when living in a specific environment is detrimental to human health and well-being (Dunn 2010; Covich et al. 2004). Potential disservices of urban gardens include invasive weeds (Mack et al. 2000, Smith et al. 2005) (Loram et al. 2008), arthropod pests (Poland and McCullough 2006), and increased exposure to trace metals (Murray et al. 2011; Sipter et al. 2008; Moir and Thornton 1989; Finster et al. 2004). Many gardens are constructed on brownfields or adjacent to automobile traffic. Reconciling trade-offs in services and disservices is a key challenge for understanding coupling between natural and human systems.

The MEA also cites evidence that degradation of the environment and subsequent reduction in ecosystem services, and increase in disservices, is borne unequally by impoverished and minority people groups. Diversity of culturally important species, natural medicines, and crop genetic diversity have declined due to agriculture homogenization, over-harvesting of species and loss of traditional ecological knowledge (Jansson and Polasky 2010). Though harder to measure, the cultural services that ecosystems provide are equally important to human well-being as ecological services. Urban gardens may serve as a medium to preserve cultural and ethnic traditions of families (especially in immigrant-run community gardens) and pass traditional knowledge along to new generations (Radford and Santos 2006).

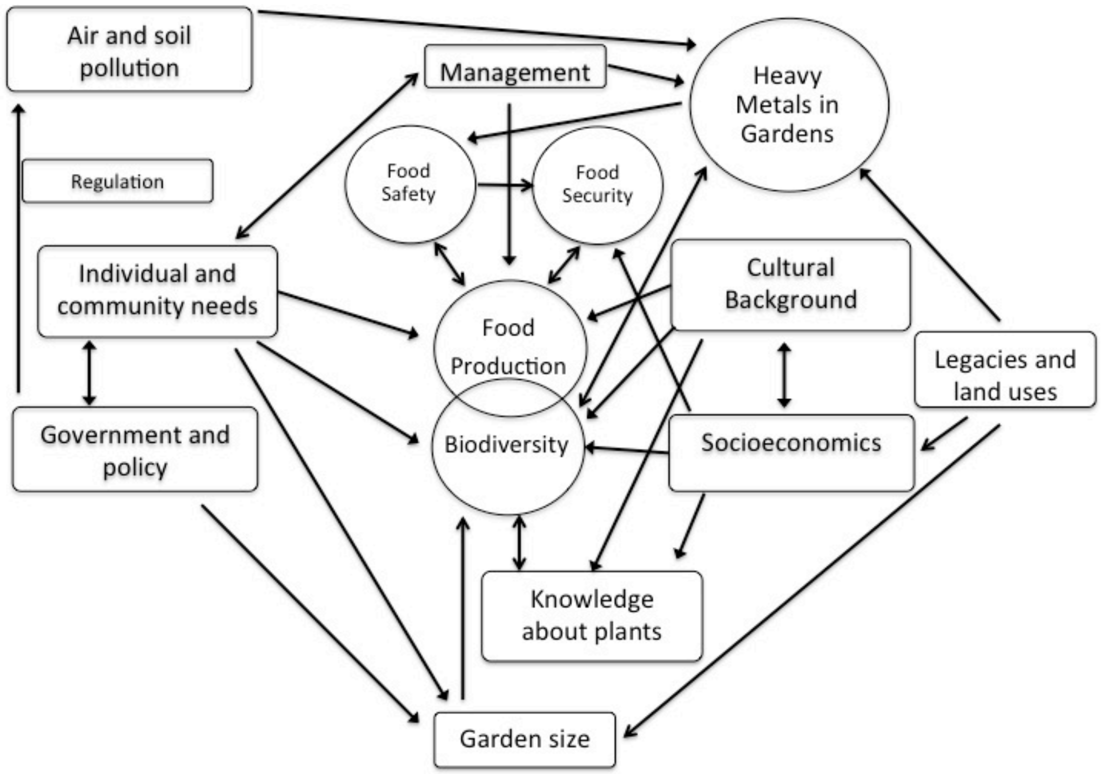

Figure 1.1: Conceptual figure showing interactions between major ecosystem services (food production and biodiversity), disservices (heavy metal contamination), and environmental, social, and cultural factors in urban gardens.

In the various projects that make up my dissertation, I mainly focus on the provisioning ecosystem services of food and medicinal production and the cultural service of aesthetics (Chapter 1 and 2). Increased ornamental biodiversity will increase aesthetics. The disservice I focus on is heavy metal contamination. Mechanisms influencing ecosystem services and disservices often overlap and interact in complex ways, as shown in Figure 1.1. In particular, both top-down (government and policy decisions) and bottom-up (individual manager and community needs) influence biodiversity and food production, either directly or indirectly (Martin et al. 2006) and feedback into each other. In home gardens, individuals may control the size of their garden, which directly influences biodiversity, but governmental restrictions and zoning may limit size of community gardens. Specifically for community gardens, while management is affected by community needs, local management decisions may influence individual needs. Also, while an individual's knowledge about plants will directly influence what species they cultivate, very biodiverse plots or neighborhood gardens may educate other gardeners, increasing overall local knowledge. Food production directly affects food security of participants, which in turn feeds back into local food production. Legacies of land use indirectly influence biodiversity and food production through spatial distribution of wealth, but directly influence disservices of heavy metals. In turn, heavy metal contamination influences human health and well-being directly and through compromising food safety, which reduces food security and directly feedbacks into food production. The amount of food being produced also dictates how much a decrease of food safety will affect participants. As heavy metals are taken up quickly by hyperaccumulators, such as brassicaceous species, and also may preclude the survival of sensitive crops, biodiversity and heavy metals affect each other.

Hierarchy of Need

One major mechanism for describing patterns of biodiversity and abundance of certain crops is that of a hierarchy of needs. This hypothesis suggests services are organized by needs progressively less connected to immediate survival. Based on this hypothesis, a luxury effect (e.g. Hope et al. 2003) is predicted where low-income gardeners focus on less diverse food species, while higher income gardeners may invest in more diversity including ornamental species (Hope et al. 2003; Peña 2005; Kinzig et

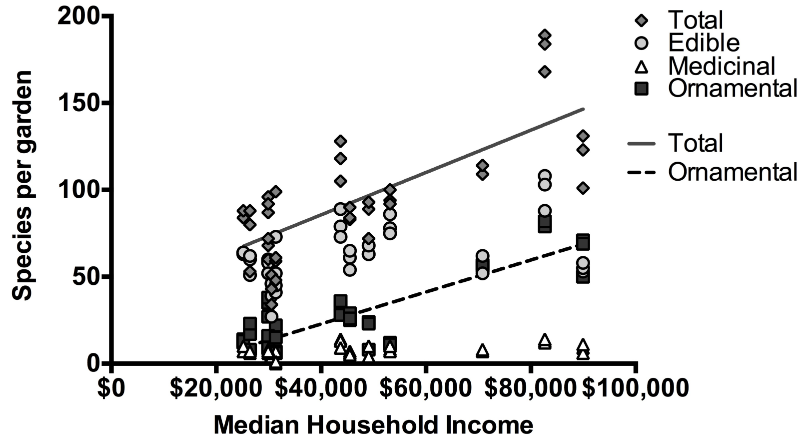

Figure 1.2: Relationships between neighborhood median income and total community garden biodiversity and major species uses (edible, medicinal, ornamental). Ornamental biodiversity is the only use that increases with income, supporting a hierarchy of needs hypothesis. This pattern is discussed more thoroughly in Chapter 1.

al. 2005; Cocks 2006). This shift from provisioning to cultural ecosystem services with increased socioeconomic status has been observed in urban gardens across the world (Thaman et al. 2006; Bernholt et al. 2009; Lubbe et al. 2010; Cilliers et al. 2012). Food security is lower in impoverished urban neighborhoods due to the presence of "food deserts," a term for an area with poor access to healthy and affordable food options (Shaffer et al. 2002). Low-income neighborhoods will then rely on the production of their garden for food, while more affluent residents have the resources to buy food from markets, allowing them to invest in aesthetic plants instead. Figure 1.2 (from chapter 1) shows this difference in edible and ornamental species with increased income.

A separate but related hypothesis, one of cultural differentiation, predicts that the set of food, medicinal, and ornamental species planted in a garden will reflect the participant's distinct cultural background. In the cultural hypothesis, individual motivations determine the ecosystem service production relationship with ecosystem dynamics -- many gardeners identify the ability to grow ethnically specific food as a reason for joining a community garden (Kingsley et al. 2009; Minkoff-Zern et al. 2012). In home gardens, the hierarchy of need may even influence the size of gardens, as more of a household's food consumption will come from the garden in impoverished areas and residents will increase the size and species density of their garden (Del Angel-Perez and Mendoza 2004; Akkinfesi et al. 2010). Food security, sufficient access to nutritional food sources (Azuma et al. 2010), may be expanded to food sovereignty in urban spaces, which is access to sufficient, culturally relevant food sources (Peña 2006). This connection ties the hierarchy of need hypothesis to the cultural differentiation one. This dissertation discusses the influence of the hierarchy of need on urban community gardens (Chapter 1) and home gardens (Chapter 2).

Heavy Metal Contamination of Urban Soils

Urban soil can be a long term sink for toxic elements, particularly heavy metal contamination from previous building materials or proximity to roads (Nicholson et al. 2002; Yesilonis et al. 2008). The large majority of metal contamination is anthropogenic, deposited by air pollution and legacies of contaminated building materials and previous land uses (Nazzal et al. 2013; Mielke et al. 1983; Charlesworth et al. 2010; Nicholson et al. 2003). The legacy of leaded gasoline additives and lead paint used in residential structures are substantial exposure risks to urban human populations (Murray et al. 2011; CDC 1985; Mielke et al. 2010). Cadmium and arsenic also build up in the soil due to anthropogenic pollution, including buildup from cadmium containing mineral fertilizers and arsenic leaching from treated wood (Yesilonis et al. 2008; Nazzal et al. 2013; Cullen and Maldanado 2013; Stillwell et al. 2006).

Urban gardens may act as an exposure route for garden participants, as metals are taken up into harvested crops (Finster et al. 2004; Säumel et al. 2012; Murray et al. 2011). Publicly managed community gardens are often established in derelict portions of the landscape with little attention paid to the presence of heavy metals (Sipter et al. 2008; Lawson 2005). Home gardeners may also be at risk, as the federal limit for lead in soils is 400 ppm (EPA 2012), though limits for where children play are much lower in CA (200 ppm) and MI (100 ppm), as there is no safe level of exposure (CDC 2013). Soil contamination may detrimentally affect food safety and, in turn, food security, impacting the production of ecosystem services (Figure 1.1). Chapter 3 covers an investigation on heavy metal presence, sources, and availability to plants in Los Angeles community gardens, addressing issues of risk and balance of ecosystem services and disservices.

Summary

Each of my chapters will address background and hypotheses for each of three major questions:

1. *What environmental and sociocultural variables influence diversity and abundance of community garden plants?*
2. *What variables influence diversity and abundance of home gardens in a developing country?*

3. *What factors influence the presence of heavy metals across community gardens in Los Angeles and how do these vary between metals of interest?*

Each of my dissertation chapters will address background and hypotheses for each of these questions. Chapter 1 covers a three-year study on community garden biodiversity in Los Angeles, CA and quantifies mechanisms influencing biodiversity patterns and production of edible, cultural, and aesthetic ecosystem services. Many community gardens in large metropolises are founded by immigrant groups, often from countries with strong home or kitchen garden traditions (Peña 2006; Gottlieb 2006). Chapter 2 covers a study completed on home garden biodiversity and ecosystem services on an urbanized gradient in Beijing, China. Here, I investigate similar mechanisms for biodiversity and ecosystem service production (cultural background, socioeconomics, size of garden) as in community gardens. The major ecosystem services obtained from urban community and home gardens are food production and cultural services (aesthetics, culturally valued food species). Heavy metal contamination in soils can affect biodiversity and viability of species being grown as well as compromising food safety for gardeners (Finster et al. 2004; Charlesworth et al. 2010). The final chapter investigates mechanisms influencing the presence and fractionation of trace heavy metals in LA community garden soils.

Works Cited

Aguílar-Stoen, M., Moe, S.R., and Lucia Camargo-Ricalde, S. 2009. Home Gardens Sustain Crop Diversity and Improve Farm Resilience in Candelaria Loxicha, Oaxaca, Mexico. Human Ecology 37(1): 55-77.

Alaimo, K., Packnett, E., Miles, R.A., and Kruger, D.J. 2008. Fruit and vegetable intake among urban community gardeners. Journal of Nutrition Education and Behavior 40(2): 94-101.

Azuma AM, Gilliland S, Vallianatos M, Gottlieb R. 2010. Food access, availability, and affordability in 3 Los Angeles communities, Project CAFE, 2004-2006. Preventable Chronic Disorders 7(2): 1-9.

Bernholt H., Kehlenbeck K., Gebauer J., and Buerkert A. 2009 Plant species richness and diversity in urban and peri-urban gardens of Niamey, Niger. Agroforestry Systems. 77:159-179.

CDC (Centers for Disease Control). Preventing lead poisoning in young children: a statement by the Centers for Disease Control. Atlanta: CDC, 1985; CDC report no. 99-2230.

CDC (Centers for Disease Control). 2012. CDC Response to Advisory Committee on Childhood Lead Poisoning Prevention Recommendations in "Low Level Lead Exposure Harms Children: A Renewed Call of Primary Prevention". Online available at: http://www.cdc.gov/nceh/lead/ACCLPP/CDC_Response_Lead_Exposure_Recs.pdf

Charlesworth, S. M. 2010. A review of the adaptation and mitigation of global climate change using sustainable drainage in cities. Journal of Water and Climate Change 1:165-180.

Cilliers, S., Siebert, S., Davoren, E., and Lubbe, R. 2012. Social aspects of urban ecology in developing countries, with an emphasis on urban domestic gardens. *In: Applied Urban Ecology: A Global Framework*, First Edition. (Richter, M. and Weiland, U. ed.) Blackwell Publishing Ltd.

Cocks, M. 2006. Biocultural Diversity: Moving Beyond the Realm of 'Indigenous' and 'Local' People. Human Ecology 34:185-200.

Colding, J., Elmqvist, T., and Olsson, P. 2003. Living with disturbance: Building resilience in social ecological systems. In *Navigating social-ecological systems:*

Building resilience for complexity and change (F. Berkes, J. Colding, and C. Folke Eds.), 163-186. Cambridge, UK: Cambridge University Press.

Colding, J., Lundberg, J. and Folke, C. 2006. Incorporating green area user groups in urban ecosystem management. Ambio 35: 237–244.

Covich, A.P., Ewel, K.C., Hall, R.O., Giller, P.S., Goedkoop, W., and Meritt, D.M. 2004. Ecosystem services provided by freshwater benthos. In *Sustaining Biodiversity and Ecosystem Services in Soils and Sediments* (ed. D.H. Wall). Island Press, Washington D.C. pp. 45-73.

Del Angel-Pérez, A.L. and Mendoza, B.M.A. 2004. Totonac homegardens and natural resources in Veracruz, Mexico. Agriculture and Human Values 21 (4): 329-346.

Diacono, M., and Montemurro, F. 2011. Long-Term Effects of Organic Amendments on Soil Fertility. In *Sustainable Agriculture, Vol 2.* (eds. E. Lichtfouse, M. Hamelin, M. Navarrete, and P. Debaeke.) Springer, Po Box 17, 3300 Aa Dordrecht, Netherlands. pp. 761-786.

Draper, C. and Freedman, D. 2010. Review and Analysis of the Benefits, Purposes, and Motivations Associated with Community Gardening in the United States. Journal of Community Practice, 18:458–492

Dunn, R. R. 2010. Global Mapping of Ecosystem Disservices: The Unspoken Reality that Nature Sometimes Kills us. Biotropica, 42: 555–557.

Fedick, S. L. (editor). 1996. The Managed Mosaic: Ancient Maya Agriculture and Resource Use. University of Utah Press. Salt Lake City, UT, USA.

Finster ME, Gray KA, and Binns HJ. 2004. Lead levels of edibles grown in contaminated residential soils: a field survey. Science and Total Environment 320:245–257

Gaston, K.J. and Gaston, S. 2011. Urban gardens and biodiversity. In: *The Routledge Handbook of Urban Ecology*. Eds Douglas, I., Goode, D., Houck, M.C. and Wang, R. Routledge, London, pp. 450-458.

Gottlieb, R. 2006. Reinventing Los Angeles; Nature and Community in the Global City. The MIT Press. Cambridge, Massachusetts, USA.

Guitart, D., C. Pickering, and J. Byrne. 2012. Past results and future directions in urban community gardens research. Urban Forestry and Urban Greening **11**:364-373.

Hope, D., Gries, C., Zhu, W. X., Fagan, W. F., Redman, C. L., Grimm, N. B., Nelson, A. L., Martin, C., and Kinzig A. 2003. Socioeconomics drive urban plant diversity.

Proceedings of the National Academy of Sciences of the United States of America 100:8788-8792.

Huai, H. and Hamilton A. 2009. Characteristics and functions of traditional homegardens: A review. Frontiers of Biology in China 47:151– 157.

Hynes, P. 1996. Patch of Eden: America's Inner City Gardens. Chelsea Green Publishing Company.

Jaganmohan, M., Vailshery, L. S., Gopal, D. and Nagendra, H. 2012. Plant diversity and distribution in urban domestic gardens and apartments in Bangalore, India. Urban Ecosystems 15: 911-925.

Jansson, A. and Polasky, S. 2010. Quantifying Biodiversity for Building Resilience for Food Security in Urban Landscapes: Getting Down to Business. Ecology and Society 15.

Kabir, M. E. and Webb, E.L. 2009. Household and homegarden characteristics in southwestern Bangladesh. Agroforestry Systems 75:129–145

Kingsley, J., Townsend, M., and Henderson-Wilson, C. 2009. Cultivating health and wellbeing: members' perceptions of the health benefits of a Port Melbourne community garden. Leisure Studies 28(2): 207- 219.

Kinzig, A. P., Warren, P., Martin, C., Hope, D., and Katti, M. 2005. The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. Ecology and Society 10(1).

Kirkpatrick J.B., Daniels G.D., and Zagorski T. 2007. Explaining variation in front gardens between suburbs of Hobart, Tasmania, Australia. Landscape and Urban Planning 79: 314-322.

Kumar, B. M., and Nair, P. K. R. 2004. The Enigma of Tropical Homegardens. Agroforestry Systems 61: 135–152.

Lawson, L. and Drake, L. 2013. Community Garden Organization Survey, 2011-2012. Community Greening Review. 18:20-41.

Loram, A., Thompson, K., Warren, P.H., and Gaston, K.J. 2008. Urban domestic gardens (XII): The richness and composition of the flora in five UK cities. Journal of Vegetation Science 19: 321–330.

Lubbe, C.S., Siebert, S.J., Cilliers, S.S. 2010. Political legacy of South Africa affects the plant diversity patterns of urban domestic gardens along a socio-economic gradient. Scientific Research and Essays 5:2900–2910.

Mack, R. N., Simberloff, D., Lonsdale, W. M., Evans, H., Clout, M., and Bazzaz, F. A. 2000. Biotic invasions: Causes, epidemiology, global consequences, and control. Ecological Applications 10:689- 710.

Martin, C. A., Warren, P. S., and Kinzig, A. P. 2004. Neighborhood socioeconomic status is a useful predictor of perennial landscape vegetation in residential neighborhoods and embedded small parks of Phoenix, AZ. Landscape and Urban Planning 69(4): 355–368.

Mathieu, R., Freeman, C., and Aryal, J. 2007. Mapping private gardens in urban areas using object-oriented techniques and very high-resolution satellite imagery. Landscape and Urban Planning 81:179-192.

Mielke, H.W., Anderson, J.C., Berry, K.J., Mielke, P.W., Chaney, R.L., and Leech, M. 1983. Lead concentrations in inner-city soils as a factor in the child lead problem. American Journal of Public Health 73 (12): 1366-1369.

Mielke, H.W., Laidlaw M.A.S, and Gonzales, C. 2010. Lead (Pb) legacy from vehicle traffic in eight California urbanized areas: Continuing influence of lead dust on children's health. Science of the Total Environment 408 (19): 3965-3975.

Michon G. and Mary F. 1994. Conversion of traditional village gardens and new economic strategies of rural households in the area of Bogor, Indonesia. Agroforestry Systems 25: 31–58.

Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: synthesis. Island press, WA, 155 pp.

Minkoff-Zern, L. A. 2012. Pushing the boundaries of indigeneity and agricultural knowledge: Oaxacan immigrant gardening in California. Agriculture and Human Values 29:381-392.

Moir, A.M., and Thornton, I. 1989. Lead and cadmium in urban allotment and garden soils and vegetables in the United-Kingdom. Environmental Geochemistry and Health 11(3-4): 113-119.

Murray, H., Pinchin, T.A., and Macfie, S.M. 2011. Compost application affects metal uptake in plants grown in urban garden soils and potential human health risk. Journal of Soils and Sediments 11(5): 815-829.

Nazzal, Y., Rosen, M.A., Al-Rawabdeh, A.M. 2013. Assessment of metal pollution in urban road dusts from selected highways of the Greater Toronto Area in Canada.

Environmental Monitoring Assessment 185:1847–1858;

Nicholson, F. A., Smith, S. R., Alloway, B. J., Carlton-Smith, C. and Chambers, B. J. 2003. An inventory of heavy metals inputs to agricultural soils in England and Wales. Science of the Total Environment **311**:205-219.

Peña, D. 2005. "Farmers Feeding Families: Agroecology in South Central Los Angeles" Lecture presented to the Environmental Science, Policy and Management Colloquium, October 10, 2005.

Peña, D. 2006. "Toward a critical political ecology of Latina/o urbanism". The Acequia Institute. Online available at: <http://www.acequiainstitute.org/researchreports.html>

Poland, T.M. and McCullough, D.G. 2006. Emerald ash borer: Invasion of the urban forest and the threat to North America's ash resource. Journal of Forestry 104:118-124.

Radford, L. and Santos, J. 2006. Race, class and the battle for South Central farm: Seeds of hope, seeds of war. *Counterpunch*. Online available at: http://www.counterpunch.org/radford07132006.html,

Säumel, I., Kotsyuk, I., Hölscher, M., Lenkereit, C., Weber, F., and Kowarik, I. How healthy is urban horticulture in high traffic areas? Trace metal concentrations in vegetable crops from plantings within inner city neighbourhoods in Berlin, Germany. Environmental Pollution 165: 124-132.

Shaffer, A. 2002. The Persistence of L.A.'s Grocery Gap: The need for a new food policy and approach to market development. Center for Food and Justice Report.

Schwarz, K., Pickett, S.T.A., Lathrop, R.G., Weathers, K.C., Pouyat, R.V., Cadenasso, M.L. 2012. The effects of the urban built environment on the spatial distribution of lead in residential soils. Environmental Pollution 163: 32-39

Sipter, E., Rozsa, E., Gruiz, K., Tatrai, E., Morvai, V., 2008. Site-specific risk assessment in contaminated vegetable gardens. Chemosphere 71: 1301-1307.

Smith R.M., Gaston K.J., Warren P.H. and Thompson K. 2005. Urban domestic gardens (V): relationships between landcover composition, housing and landscape. Landscape Ecology 20: 235-253.

Stark, B.L., and Ossa, A. 2007. Ancient settlement, urban gardening, and environment in the Gulf Lowlands of Mexico. Latin American Antiquity 18:385-406.

Thaman, R.R., Elevitch, C.R., and Kennedy J. 2006. Urban and homegarden agroforestry in the Pacific Islands: Current status and future prospects. In: *Tropical homegardens: a time-tested example of sustainable agroforestry. advances in agroforestry*, Kumar B.M., Nair P.K.R. (eds) vol 3. Springer, Dordrecht, pp 25–41

Yesilonis, I., Pouyat, R., and Neerchal, N.K., 2008. Spatial distribution of metals in soils in Baltimore, Maryland: role of native parent material, proximity to major roads, housing age and screening guidelines. Environmental Pollution 156 (3): 723-731.

Zhu, W., Hope, D., Gries, C., Grimm, N.B. 2006. Soil characteristics and the accumulation of inorganic nitrogen in an arid urban ecosystem. Ecosystems 9:711–724.

Chapter 2: Regulation of the Extensive Biodiversity and Direct Ecosystem Services in the Community Gardens of Los Angeles, CA

Abstract

Urban community gardens, globally prevalent urban agricultural areas, have the potential to fulfill ecosystem needs in impoverished neighborhoods, such as food security and biodiversity. Despite these benefits, little research has been conducted that evaluates environmental and socioeconomic factors influencing community garden plant biodiversity and ecosystem services. My study investigated the drivers of managed plant richness, abundance, and direct ecosystem service production in 14 community gardens across Los Angeles County, CA between 2010-2012. The investigation spanned regional, garden, and plot scales, identifying scaling relationships in biodiversity patterns. In total, 707 managed species were recorded in summer surveys over a three-year period. Ornamental richness increased with neighborhood income, while food and medicinal richness was positively related to size of garden plots. Gardener cultural background also influenced the composition of managed species, especially edible species. I explain these patterns through a hierarchy of needs and cultural preference framework. Ornamentals are luxury species; purchasing and maintaining a variety of flowering species takes time and resources and does not produce more essential provisioning services. Edible species production may moderate food insecurity in low-income neighborhoods, where ornamentals are used less often. Culturally specific crops provide both cultural and provisioning ecosystem services for immigrant populations, resulting in culturally distinct

assemblages of edible species. Finally, demand for food abundance and biodiversity is not met in small garden plots, creating a distinct species-area relationship. These quantitative results indicate that community gardens contribute to a bio-diverse urban ecosystem and provide valued ecosystem services in food insecure regions.

Introduction

Community gardens are increasingly recognized as a source of food security for low-income residents with broad support from local, state and federal agencies (Irazabal and Punja 2009; Gottlieb 2006; Foltz et al. 2012). These agricultural spaces run by groups of residents are becoming more prevalent in southern California and the United States with over 3000 community gardens established between 2007-2011 (Lawson and Drake 2013). They are important sources of direct and indirect ecosystem services (ES), directly benefitting participants through edible and medicinal crop production (Smith et al. 2013) and aesthetics (Peña 2005, Ailamo et al. 2008), as well as indirect ES, like aiding neighborhood pollinators (Matteson et al. 2008), neighborhood cooling (Jenerette et al. 2011), and pollution reduction (Manes et al. 2012). Despite these recorded benefits and increased interest, there have been no studies quantifying the potentially large community garden cultivated biodiversity, direct ES produced by these plants, or their relationship to hypothesized mechanisms (Guitart et al. 2012). My study addresses this knowledge gap, focusing on how garden biodiversity and ecosystem services change with the needs and values of residents from different socioeconomic and cultural backgrounds.

The organization of community gardens makes them a unique multi-scale system to explore the coupling between diverse human and biophysical systems in a highly structured and replicated setting. As a Coupled Human and Natural System (CHaNS, Liu et al. 2007), community gardens feature extensive social and biological diversity, predictable changes at regular intervals, well-defined boundaries of individual sub-systems within nested hierarchies, and widespread replication within gardens and cities. The management of community gardens may be more individually focused, with each plot benefitting a single family, or more communally focused, like in farms, where production across plots is shared between multiple participants (Jackson et al. 2013). While each garden has a single management style (i.e. individual gardens or farms), each plot is individually maintained for species selection, soil preparation, and applications of fertilizers and irrigation. Spatial and temporal variation in community garden biodiversity and ES production arises from a combination of biotic, abiotic, and human processes. The organization of between 10-150 individual plots within multiple gardens allows the quantification of biodiversity at three different ecological scales that together provide information on the amount and structure of diversity (Anderson et al. 2011): α (alpha diversity: individual plot scale), γ (gamma diversity: whole garden scale) and β diversity (turnover between plots in a single garden). Variation in biodiversity across these scales may be influenced by a variety of interacting factors including organization, neighborhood income, culture, ecosystem services, and planting area.

One framework to better understand the interaction of income, culture, and direct ES is a hierarchy of needs, where services are organized by needs progressively less connected to immediate survival (Lubbe et al. 2011, Clarke et al. 2013, Wu 2013, Kinzig et al. 2005, Cocks 2006). Financial resources and leisure time necessary for investment in garden maintenance is dependent on the socioeconomic status of individual gardeners (Pickett et al. 2011, Lawson and Drake 2013). In large metropolises, median family income varies widely across neighborhoods and city regions, affecting both garden resources and species biodiversity (Jackson et al. 2013). Low-income gardeners may have unmet nutritional and culturally specific food needs that focus their output on edible species, while higher income gardeners may have their food needs met commercially and therefore select more ornamentals that fulfill aesthetic desires (Gaston and Gaston 2011, van Heezik et al. 2013).

Alternatively, a cultural hypothesis predicts that the set of food, medicinal, and ornamental species planted in a garden will be distinct to the participant's cultural background. Though all gardeners may share the same basic needs for ES (food, aesthetic beauty, medicines), the palette of species valued for food or aesthetics varies between cultures (Fraser and Kenney 2000; Kinzig et al. 2005, Wakefield 2007). Large variations in cultural diversity and high immigrant participation in gardens across urban regions may also potentially contribute to proliferation of culturally specific crops in gardens (Wakefield et al. 2007, Gottlieb 2006). In addition to cultural preferences, immigrant gardeners may also be more likely to come from agricultural regions that have strong gardening traditions (Minkoff-Zern 2012), and may possess more skill in maintaining diverse gardens (Barthel et al. 2010).

Biodiversity patterns may also be linked to production of ES demanded by gardeners. Crops that supply culturally significant provisioning services, like food or medicine, may be more valued for abundance and productivity than biodiversity (Cilliers et al. 2012). If multiple gardeners in a single garden value the same species for its food production due to its cultural importance, edible β diversity may be low. In contrast, gardeners may plant a variety of unique ornamental species in order to express individual tastes (Kaplan and Herbert 1987, Marco et al. 2008), thereby creating extensive aesthetic β diversity. Higher β diversity through individual species choice may impact biodiversity shifts between years. A legacy hypothesis predicts that older gardens will be more bio-diverse than more recently established gardens. Older, well-established gardens with secure tenure may steadily become more bio-diverse over time through legacies of species from previous managers, similar to legacy patterns observed across entire cities (Pickett et al. 2011, Clarke et al. 2013) and in residential yards (Larsen and Harlan 2006).

Separate from socio-cultural influences, a fundamental ecological relationship explaining changes in species diversity is the species-area relationship (Lawton 1999, Koellner and Schmitz 2006). Area in community gardens varies at two scales: whole gardens and individual plots. Though studies across the world have shown a positive relationship between domestic garden size and species biodiversity (Smith et al. 2005, Loram et al. 2008, Huai et al. 2011), this relationship is not always observed (Albuquerque et al. 2005, Clarke et al. 2014). In community gardens, I hypothesize the species-area relationship will be influenced by garden management styles (individually-based vs. farms) and vary between scales. Species area relationships have been shown in subsistence gardens that support individual families (Méndez et al. 2001, Kabir and Webb 2009), though not in larger farms (Blanckaert et al. 2005). In communally-based farms, garden production is shared between participants and marketable species are often sold or donated, as in church or school gardens. The modified species-area hypothesis predicts that garden species diversity will be linked to plot size in individually-based gardens if demand for provisioning or aesthetic ES exceeds space available for desired species.

This study aims to answer basic questions about community garden biodiversity, abundance, and direct ES production. I ask, *how diverse are community garden cultivated floras and what factors regulate the biodiversity and distribution of garden plants?* I also ask, *what factors influence species choice and direct ecosystem service production in gardens?* To answer these questions, I investigated temporal and spatial-scale variation of biodiversity and ecosystem service production across fourteen community gardens in Los Angeles, CA for three years. I expect interactions between different mechanisms affecting biodiversity and direct ES– garden management style, socioeconomics, cultural identity, species-area relationships– will create complex patterns of vegetation diversity and direct ES production.

Methods

Study area

The socio-ecological heterogeneity of Los Angeles (LA) provides a useful site to study variability among community gardens. Over 30% of LA County's population is foreign-born, with 45% of the population of Hispanic descent (2010 U.S. Census Bureau). Neighborhood median household income ranges widely from $9,000-200,000. Low-income neighborhoods in LA have some of the highest immigrant and minority concentrations in the entire U.S. (U.S. Census 2010). These impoverished neighborhoods have only 1 grocery store per 46,000 residents, as compared to 1 store per 20,000 residents in higher income neighborhoods (Schaffer 2002). There are 99 officially recognized community gardens across LA, 60% of which are set in low-income neighborhoods with high immigrant populations (Figure 2.1). Throughout the 1980s and 1990s, most LA community garden projects were either initiated or expanded by Latino immigrants (Gottlieb 2006).

Field methods

Beginning in 2010, I selected 14 community gardens within Los Angeles County for inclusion in this study. These gardens are located in neighborhoods with median incomes between $25,000-$90,000, range in size between 400-10,000 m^2, and were established between 1963-2009 (Table 2.1). Of these, seven gardens included mainly Hispanic immigrants primarily from Mexico, but also from Guatemala, El Salvador, and Costa Rica. One garden had a majority of Korean immigrants. Together, these 8 community gardens were categorized as "immigrant" gardens. The remaining 6 had a majority of U.S. born residents, and were categorized as "non-immigrant" gardens. Of these, one garden was made up exclusively of African-Americans from the American Southwest. Each garden was also categorized by management style. Nine gardens were identified as individually-based gardens, where 1-2 participants manage small (~4.5-50 m^2) plots and the produce is not sold or used to support multiple families. I also sampled five farms, defined as communally-based gardens with large (~50-135 m^2) plots, monocultured rows, shared crop production, and selling of produce for profit.

The area of each whole garden was measured using Google Earth and the size of each discrete plot was measured on site. Garden managers provided information about date of establishment and history of the garden. Median income was estimated for each garden neighborhood using the neighborhood census data from 2010 compiled by the LA Times (http://projects.latimes.com/mapping-la/neighborhoods). Cultural identity of gardens was determined by talking to managers and observation.

Comprehensive species presence and abundance inventories were completed in each individually owned plot and for the whole garden (including common areas) during summers of 2010-2012. Each garden was visited and surveyed once each year between the months of June-August. All deliberately cultivated plants were identified and percent cover of each species estimated based on visual inspection. Covers were grouped into five area categories (0-5% (Rare); 5-25% (Uncommon); 25-50% (Common); 50-75% (Very Common); 75-95% (Abundant); 95-100% (Dominant)). I then estimated m^2 of each species in a plot by taking the midpoint proportion of each category and multiplying that by plot size. As some plots had multiple layers of crops, this technique allowed the area of crops in a plot to be >100%.

Species, not varieties, were recorded with a few exceptions. If different parts of the plant were used or one variety provided a separate use, they were recorded separately.

For instance, *Brassica oleracea* encompasses a variety of distinct food products, such as broccoli, collards, and kohlrabi, each of which were recorded separately. In contrast, yellow crookneck squash and zucchini (both *Cucurbita pepo*) were only recorded as a single species as this difference did not result in variation of plant parts. Gardeners were asked about the identity of unknown species. Proper taxonomic identification for unusual species was assured through photos and collection of voucher specimens for expert identification and archiving at the UC Riverside herbarium. I divided species into broad use categories based on whether the species provided various provisioning or aesthetic/cultural ES. These categories included edibles (E) and medicinals (M), both provisioning uses, and ornamentals (O), plants with cultural or aesthetic service value. In addition, I include an "Other" category (D) for less common provisioning and cultural services. Other included plants used for spiritual purposes (e.g. *Tagetes erecta* used in *Dia de los muertos*), fiber plants, shade trees, and pest deterrents. Many plants had multiple uses, so the sum of edible, medicinal, ornamental, and other species was greater than total richness. The most common species in each use can be seen in Appendix 1.1.

Data analysis

Each distinct garden/year combination was used as a unique data point in these analyses and I further examined garden patterns in each year and then patterns of individual and conglomerated gardens across years. This helped me identify how patterns of biodiversity and ES production varied across and within years. I used both one-way ANOVA, for comparison of abundance of different uses across management styles and immigrant status, and linear regressions to examine controlling factors on ecological variables (SPSS 11.3).

To test for any correlations between my main hypothesized mechanisms, I conducted a Pearson's product moment correlation between garden age (years since establishment), plot size (m^2), and median neighborhood income for all gardens and separately for individually based and farm managements. Garden age was adjusted for each sequential year (e.g. a 20 year old garden in 2010 was recorded as 21 in 2011) and plot size was re-measured each year. As the most recent Census was completed in 2010, median neighborhood income remained the same across time. I found that for individually-based gardens, plot size was positively correlated with both age of garden and neighborhood income (Table 2.2). The age-size correlation is unsurprising, as gardens built before the 1980s were established before a major housing boom in Los Angeles and more open space was available for garden plots (Gottlieb 2006). In addition, income and population density are negatively related across Los Angeles (Clarke et al. 2013, Census 2010), so affluent neighborhoods may accommodate larger gardens and plot sizes. To account for these co-linearities, I conducted a multiple regression to determine which combinations of factors were influencing each biodiversity or abundance measurement. Multiple regressions were conducted separately for total number of plot species, average number of species per plot, and species abundance and repeated for each different use, immigrant status, and management style. When multiple regression models indicated that a combination of two or more variables had a detectable effect on biodiversity or abundance, I used a controlled regression to examine individual variable effects. For this, each significant variable identified in the multiple regression was regressed against the residuals of a regression on the biodiversity or abundance measure and the other identified variables.

I used the Jaccard's index to determine β diversity or turnover between plots in a single garden in a single year (Anderson et al. 2011). Matrices of species presence-absence were used to compare biodiversity across all plots in the same garden (EstimateS 9.0). Resulting values were inverted to create an average Jaccard's dissimilarity index for each garden. This analysis was repeated for edibles and ornamentals in each garden and then the combination of three years was compared between uses with an ANOVA.

Non-metric multidimensional scaling ordination (NMDS) of the Jaccard's dissimilarity metric was used to analyze species turnover rates between garden sites (Anderson 1971, Cilliers et al. 2012). This ordination is nonlinear, and creates a physical representation maximizing distance based on rank-order agreement with their dissimilarities in species composition (Austin 2005). A Jaccard's dissimilarity matrix was created from a species presence-absence matrix (EstimateS 9.0). This matrix compared each garden in each year to all other gardens in all other years. The ordination was then projected it in two dimensions (PROXSCAL on SPSS). This analysis was repeated using only edible or ornamental matrices. I then divided gardens into culturally distinct groups (as labeled in Table 2.1) in order to determine whether cultural differences influenced species composition. Differences between ethnic groups were tested using a one-way ANOVA on each ordination axis.

Results

Biodiversity patterns

Overall, biodiversity was high across all gardens, with 707 species identified in garden plots across the three years of the study (Table 2.3). Over half the species were ornamental, with the four non-immigrant individual gardens containing the highest ornamental richness (185 species) and highest overall species richness (349 species) (Table 2.3). Though ornamentals had a higher biodiversity than edible species when combined across multiple gardens, a t-test indicated that edibles outnumbered ornamentals in each garden (γ) by a factor of three (Figure 2.2a; $p<0.001$) and by a factor of four for plot (α) diversity (Figure 2.2b; $p<0.05$). The exception to the pattern was a single non-immigrant farm in the highest income neighborhood, which had more ornamentals than edibles at the α and γ scale (NIMM5). Species in plots were correlated with the number of species in each garden ($r^2=0.53$, $p<0.001$; Figure 2.3), a pattern repeated for edible and ornamental species, indicating that high α biodiversity influenced larger scale γ biodiversity. In addition, I found no consistent temporal pattern for abundance or species richness across all gardens, with individual gardens increasing, decreasing, or having consistent biodiversity (Figure 2.2).

Socioeconomics and species uses

Multiple regression indicated that both neighborhood income and garden plot size were related to overall species richness. Controlled regressions then indicated that both neighborhood income and plot size had independent effects on species richness. Species biodiversity and cover were significantly related to neighborhood income in regressions controlling for the effect of plot size, though patterns differed between uses (Figure 2.4A,B). Overall species richness was related to income ($r^2 = 0.468$, $p<0.001$), but only ornamentals increased with income ($r^2 = 0.620$, $p<0.001$). Ornamental cover was also positively related to income ($r^2 = 0.530$, $p<0.001$). Edible and medicinal species richness and cover showed no significant relationship with income ($p>0.05$). Though non-immigrant gardens were the only ones with a significant income-diversity relationship, immigrant-run gardens are located primarily in low-income neighborhoods. This indicates that their reduced income may be obscuring the observed overall luxury effect.

Ornamental α and γ biodiversity were lower than edible species within and between gardens, but had a consistently higher turnover rate than edibles (β) (Figure 2.5, Table 2.4). In each sample year, about 60% of identified ornamentals were found in less than 1% of garden plots, and no ornamental species were planted in more than 10% of garden plots. In contrast, while 40% of edibles found in each year were also found in less than 1% of garden plots, they were more evenly distributed across plots. Many edible species were found in 20-35% of all plots. Within a single garden, β diversity varied between uses. While overall β was high between individual plots within a garden (Jaccard's dissimilarity >0.8), ornamental β was the highest ($p<0.01$; Table 2.4).

Individual vs. communal-based (farm) management style and immigrant status of community gardens affected the overall cover patterns (Figure 2.6). While individual based garden plots had similar edible cover in both immigrant and non-immigrant locations, immigrant farms had the highest edible cover ($p<0.01$). Ornamental cover was highest in non-immigrant gardens and conversely lowest in immigrant farms ($p<0.001$), while medicinal cover was the highest in immigrant gardens. In addition, edible cover was higher than ornamental across all gardens, ranging from 40-140% in each plot, while ornamentals ranged from 1-30% (Figure 2.6). Ornamental and edible cover were both related to their respective species richness, though edible explanatory value was low (Ornamental: $r^2= 0.68$, $p<0.01$; Edible: $r^2= 0.14$, $p<0.05$).

Cultural background

NMDS for all gardens indicated that cultural background influenced species composition within and across species uses. For all species (Figure 7A-1), predominantly Hispanic/Asian gardens had significantly different compositions than predominantly African-American/Non-immigrant gardens, and were oriented in a unique location in axis 1 (Figure 2.7-A-2). For edible species (Figure 2.7-B-1), Hispanic gardens had a different set of species than all other gardens, most clearly shown on axis 1 (Figure 2.7-B-2), and African-American gardens differed from others on axis 2. Finally, for ornamental species, plant distributions were more variable, though Hispanic gardens included significantly different species than non-immigrant gardens (2.7-C-1, C-2). Cultural garden groups were most similar in food species, while ornamental species were the most dissimilar (Figure 2.8). Individually, gardens were self-similar across the three years of the study (average Jaccard's dissimilarity: 0.5) and the most dissimilar between gardens of different cultural backgrounds in the same years (0.7; Figure 2.8).

Species-area relationships and legacies

Garden scale species richness was positively related to size of individual plots (r^2=0.214; p=0.01; Figure 2.9A), when controlled for income, but only in individually-based gardens, not farms (Figure 2.9B). For individually-based gardens, the species-area relationship was the most evident for edible species (r^2=0.221; p=0.009), with no effect of income on edibles. Ornamental species richness was unrelated to size. Farm-style gardens had low variation in the number of species found within gardens, regardless of plot size, a pattern that remained the same across all species uses. Age and size were both shown as controlling variables in stepwise regression for medicinal species. When controlled for the effect of plot size, age of garden had no independent effect on the number of species per garden (r^2=0.02, p=0.23). I compared species uses, immigrant status, and garden management separately in the controlled regression, and none showed any abundance or biodiversity relationship to garden establishment date.

Discussion

Los Angeles community gardens are hotspots of biodiversity, with over 700 managed species in a total area of only 6.5 ha, or nearly 100 species per hectare across three years. Considering that this subsample of community gardens represents less than 20% of the 100 gardens in Los Angeles County, the number of regularly managed species in Los Angeles gardens may be higher than previous studies of entire metropolises (Walker et al. 2009). This high biodiversity and the ES provided in Los Angeles community gardens are driven by a combination of garden management, income, cultural identity, and area. Scale-specific variation of biodiversity and frequencies of species are linked to ES provided and garden management style (Figure 2.5, 2.6). No legacy effect was detected, as some gardens increased in biodiversity or abundance across years, some decreased, and others fluctuated with no clear pattern (Figure 2.2). These findings support a hierarchy of need coupled with cultural preferences, indicating that impoverished immigrant gardeners will focus on culturally important food species (Figure 2.7, 2.9), while high-income gardeners have increased resources to invest in ornamental diversity (Figure 2.4). I also found that species-area relationships exist only at the plot scale in individually-based gardens, primarily influencing edible species (Figure 2.9A,B); thus indicating management style and ES influence space demands. My interdisciplinary project links socioeconomic, cultural, and spatial scale patterns to biodiversity and species uses in community gardens, quantifying production of direct ES.

Socioeconomics and the hierarchy of need

Species uses and ES production in community gardens are related to median family income (Figure 2.4), supporting a hierarchy of need hypothesis (Wu 2013). In Los Angeles, a "food desert" (inadequate access to grocery stores) exists in low-income neighborhoods and contributes to reduced food security (Azuma et al. 2010). These results are consistent with low-income garden participants responding to reduced access to resources by selecting crops that provide edible ES, and not investing in ornamentals (Figure 2.4, 2.7). Food crops may improve gardener livelihoods through providing basic food needs (Ailamo et al. 2008, Gatto et al. 2012, Clarke et al. 2014). Ornamental richness in affluent neighborhoods may be attributed to luxury investments in aesthetic and cultural ES and decreased demands for edible species. Heterogeneity of ornamental species (Table 2.4) is likely present as a result of affluent gardeners expressing preferences through unique ornamentals (Marco et al. 2010). This shift from provisioning to cultural and aesthetic ES with increasing socioeconomic status has been observed in cities across the world (Hanna and Oh 2000, Kinzig et al. 2005, Loram et al. 2008). While edible species richness does not decrease with increasing income, higher income gives gardeners resources to invest in flowering species (Cilliers et al. 2012) and intensively manage more extensive plant assemblages (Walker et al. 2009, Lowry et al. 2012).

Patterns of scale-specific spatial variation may also be interpreted using a hierarchy of need. Regional and garden scale richness display different patterns in allocation of species providing ES. Though ornamentals outnumber edibles regionally, each garden has more edible richness (Table 2.1). This pattern is associated with differences in β diversity, as 1/3 of edible species are planted in 15-30% of all plots and no ornamental species are planted in more than 10% of plots (Figure 2.5). If a gardener sets aside increased space for ornamental abundance, then the tendency is to increase variety of ornamentals, while gardeners are more likely to increase abundance of a few edible species if edible area is increased (Figure 2.5). Gardener valuation of provisioning and aesthetic ES may explain the proportional difference. The high β diversity in garden scales (Table 2.4) indicates ornamentals are valued for their variety and "difference" (Marco et al. 2010). Specific food needs fulfilled by each edible species are not fulfilled by diversity and gardeners may instead value a few food species to sustain their family (Galluzzi et al. 2010, Hale et al. 2011).

Cultural preferences

Culturally distinct groups of gardeners grow distinctly different sets of garden species (Figure 2.7). Edibles in particular show more cultural distinction than other plant uses (Figure 2.8). Consistent with these spatial patterns, gardens were also consistently similar in species biodiversity, especially edibles, across multiple years (Figure 2.8). Both the spatial and temporal patterns are consistent with culturally expressed values for food sovereignty. Cultivating culturally relevant plants helps immigrants maintain cultural identity and agrarian traditions in an unfamiliar environment (Corlett et al. 2003, Peña 2006).

Edible species in immigrant gardens may express social heritage and history in culturally important food sources (Fu et al. 2006; Hale et al. 2011). Many immigrant participants indicate a desire for fresh, familiar produce in their gardens (Corlett et al. 2003). Though ornamental composition is less segregated to culture than edibles (Figure 2.7C, Figure 2.8), it also contains a cultural component. For instance, *Tithonia rotundifolia* and *Tagetes erecta* are both used as ornamental species in Hispanic gardens. But they also provide important cultural services, and are used extensively in the *Dios de los Muertos* celebration throughout Central America. Americans, Europeans, Hispanics, and Asians can have very different preferences for decorative landscapes (Kaplan and

Herbert 1987, Fraser and Kenney 2000, Kinzig et al. 2005), which may explain some of the cultural preferences in ornamental choice.

Garden area

Garden management style and local species preference affected species-area relationships across community gardens for plot size, not garden size, affecting edible and medicinal biodiversity only in individually-based gardens (Figure 2.9A). Similarly, in England and New Zealand home gardens, species richness was most influenced by individual garden size, not the overall amount of garden space in a community (Smith et al. 2005, van Heezik et al. 2013). A consistently linear species-area relationship in plots indicates garden space is insufficient to support demand for crops producing direct ES (Loram et al. 2008, Kabir and Webb 2009, Van Heezik et al. 2013). The mechanism behind management specific species-area relationships may be based on the hierarchy of need. Farms often share food communally, so there is less pressure for a single plot manager to grow all edibles necessary for sustenance (Pedro Barrera, farm manager, pers. comm). In individually-based gardens, participants who desire a certain suite of species must grow them all in a single plot. With increased space, more species are planted to address ES demand, leading to the observed species-area relationship, a pattern also described in home gardens (Albuquerque et al. 2005, Loram et al. 2008, Clarke et al. 2014). In contrast, ornamentals take up a much smaller area of the garden (Figure 2.6) and my other results indicate they are valued for diversity, not cover (Figure 2.5, Table 2.4). Species abundance patterns are also affected by both management and immigrant status (Figure 2.6). Gardeners, like those in farms, who rely monetarily on garden success may be more likely to plant edible species because of their commercial value (Fu et al. 2006, Lubbe et al. 2011, Galluzzi et al. 2012). This pattern is evident in immigrant farms, which have the highest abundance of edibles and conversely lowest ornamentals.

Garden age

The results do not show a legacy effect of garden age on species biodiversity patterns. Previous studies showing a clear effect of age of development on biodiversity were from surveys of trees or perennials, which are uncommon in community gardens (Loram et al. 2008, Boone et al. 2010, Clarke et al. 2013). I had initially posited that older gardens could indicate high land tenure and security for gardeners, encouraging crop legacies. This may indicate that the age of gardens may be a poor proxy for gardener tenure and security. Further studies incorporating individual gardener decisions about tenure may better evaluate legacy effects.

Synthesis and Extensions

My research demonstrates the extensive biodiversity of urban community gardens and quantifies direct ES benefits for participants. These findings support the proposition that urban agriculture, like community gardens, can contribute to food security and cultural expression in cities (Chappell and LaValle 2011). These highly diverse and dynamic crop repositories may be considered a secondary Vavilov center of global biodiversity (Vavilov 1949), where high genetic biodiversity in LA is being created and maintained by gardeners imposing selection pressure on crop species over multiple years (Heraty 2010; Soleri and Cleveland 2004). In addition to the direct services, I further expect that high biodiversity can also support indirect ES, like pollination and pollution reduction. Further, potential disservices of urban agriculture, like weed proliferation or encouragement of pest species, should be evaluated to better understand and minimize ES trade-offs associated with urban agriculture. The results of this intensive study provide comprehensive information on drivers of community garden biodiversity, abundance and ES production in a large and diverse U.S. metropolis – community garden biodiversity is influenced interactively by income, culture, management, and area. My results offer decision support for planning urban gardens as multifunctional green spaces (Lovell and Taylor 2013). Plot size is an important trade-off in garden design and will influence the ability of individual gardens to fully meet ecosystem service needs. As part of reaching impoverished residents, providing access to culturally diverse seeds helps maximize individual benefits. This quantitative data helps "close the loop" in linking gardener and societal desires to ecosystem service production (Lawson 2007; Gottlieb 2006).

More broadly, I suggest community gardens are a model system for better understanding of human-ecosystem functioning related to biodiversity and the production of ES. While the mechanisms I propose in this paper are of broad application individually (hierarchy of need, cultural preferences, size of plot), they interact with each other significantly. For example, most of the immigrant gardens were located in impoverished neighborhoods, which may be an example of more general interactions between ES needs. These causes of variation and their interaction may be broadly applicable in

CHaNS where ecosystem services are regulated by both social and environmental heterogeneity.

Conclusion

My results identified both influences of a hierarchy of need and cultural specificity in shaping both community garden biodiversity patterns and production of edible, cultural, and aesthetic ES. Edible species are planted based on cultural background and demand for food production in impoverished neighborhoods, while ornamental species are planted in affluent neighborhoods for "luxury" aesthetics and is less connected to cultural preferences. This shift from aesthetic to provisioning ES with reducing income is reflective of the hierarchy of need; with decreased financial resources, food becomes a priority. I also found that the management and size of gardens affect planting patterns, with unmet demand for species diversity in smaller plots leading to species-area relationships. As community gardens are proliferating across the country (Corrigan 2011; Lawson and Drake 2013), the hierarchy of need results indicates demand for policy makers to create more secure, accessible gardens for participants in lower income neighborhoods. Community gardens contribute to a bio-diverse urban ecosystem and maintaining a secure supply of culturally relevant food plants in food-insecure regions.

Works Cited

Alaimo, K., Packnett, E., Miles, R.A., and Kruger, D.J. 2008. Fruit and vegetable intake among urban community gardeners. Journal of Nutrition Education and Behavior 40(2): 94-101.

Albuquerque, U. P., Andrade, L. H. C., and Caballero J. 2005. Structure and floristics of homegardens in Northeastern Brazil. Journal of Arid Environments 62:491-506.

Anderson, M. J. 2011. Navigating the multiple meanings of beta diversity: a roadmap for the practicing ecologist. Ecology Letters 14:210-210.

Anderson, A. J. B. 1971. Ordination methods in ecology. Journal of Ecology 59: 713-72.

Austin, M.P. 2005. Vegetation and environment: discontinuities and continuities. *In Vegetation Ecology* (van der Maarel, E., ed.), pp. 52–84, Blackwell Publishing, Ltd.

Azuma, A.M., Gilliland, S., Vallianatos, M., Gottlieb, R. 2010. Food access, availability, and affordability in 3 Los Angeles communities, Project CAFE, 2004-2006. Preventable Chronic Disorders 7(2): 1-9.

Barthel, S., Folke, C., and Colding, J. 2010. Social-ecological memory in urban gardens-Retaining the capacity for management of ecosystem services. Global Environmental Change-Human and Policy Dimensions 20:255-265.

Blanckaert, I., Vancraeynest, K., Swennen, R.L., Espinosa-Garcia, F.J., Piñero, D., and Lira-Saade, R. 2007. Non-crop resources and the role of indigenous knowledge in semi-arid production of Mexico. Agriculture Ecosystems and the Environment 119(1-2): 39-48.

Boone, C. G., Cadenasso, M. L., Grove, J. M., Schwarz, K., and Buckley, G. L. 2010. Landscape, vegetation characteristics, and group identity in an urban and suburban watershed: Why the 60s matter. Urban Ecosystems 13(3): 255–271.

Chappell, M. J. and LaValle, L. A. 2011. Food security and biodiversity: can we have both? An agroecological analysis. Agriculture and Human Values 28(1), 3-26.

Cilliers, S., Siebert, S., Davoren, E., and Lubbe, R. 2012. Social aspects of urban ecology in developing countries, with an emphasis on urban domestic gardens. *In: Applied Urban Ecology: A Global Framework*, First Edition. (Richter, M. and Weiland, U. ed.) Blackwell Publishing Ltd.

Clarke, L. W., Jenerette G.D., and Davila, A. 2013. The luxury of vegetation and the legacy of tree biodiversity in Los Angeles, CA. Landscape and Urban Planning 116:48-59

Clarke, L.W., Li, L., Jenerette, G.D., Yu, Z. Drivers of plant biodiversity and ecosystem service production in home gardens across the Beijing Municipality of China. Urban Ecosystems DOI 10.1007/s11252-014-0351-6.

Cocks, M. 2006. Biocultural Diversity: Moving Beyond the Realm of 'Indigenous' and 'Local' People. Human Ecology 34:185-200.

Colding, J., Lundberg, J. and Folke, C. 2006. Incorporating green area user groups in urban ecosystem management. Ambio 35: 237–244.

Colding, J., Elmqvist, T., and Olsson, P. 2003. Living with disturbance: Building resilience in social ecological systems. In *Navigating social-ecological systems: Building resilience for complexity and change* (Berkes, F., Colding, J., and Folke, C. Eds.), 163-186. Cambridge, UK: Cambridge University Press.

Colding, J., Barthel, S., Bendt, P., Snep, R., van der Knapp, W., and Ernstson, H. 2013. Urban green commons: Insights on urban common property systems. Global Environmental Change. 23(5) 1039-1051.

Corlett, J., Dean, E., Grivetti, L., 2003. Hmong gardens: botanical diversity in an urban setting. Economic Botany 57: 365–379.

Corrigan, M.P. 2011. Growing what you eat: developing community gardens in Baltimore, Maryland. Applied Geography 31: 1232–1241.

Draper, C. and Freedman, D. 2010. Review and Analysis of the Benefits, Purposes, and Motivations Associated with Community Gardening in the United States. Journal of Community Practice 18:484-492.

Fraser, E. D. G. and Kenney, W.A. 2000. Cultural background and landscape history as factors affecting perceptions of the urban forest. Journal of Arboriculture 26(2):106-113.

Fu, Y., F., Guo, H., Chen, A., and Cui J. 2006. Household differentiation and on-farm conservation of biodiversity by indigenous households in Xishuangbanna, China. Biodiversity Conservation 15:2687–2703.

Galluzzi, G., Eyzaguirre, P., and Negri, V. 2010. Home gardens: neglected hotspots of agro- biodiversity and cultural diversity. Biodiversity and Conservation 19(13): 3635-3654.

Galluzzi G. 2012. Agrobiodiversity and protected areas: another approach to synergies between conservation and use? In: *On-farm conservation of neglected and underutilized species: status, trends and novel approaches to cope with climate change* – Proceedings of the International Conference, Friedrichsdorf, Frankfurt, 14-16.

Gaston, K.J. and Gaston, S. 2011. Urban gardens and biodiversity. In: *The Routledge Handbook of Urban Ecology*. Eds Douglas, I., Goode, D., Houck, M.C. and Wang, R. Routledge, London, pp. 450-458.

Gatto, N. M., Ventura, E. E., Cook, L.T., Gyllenhammer, L. E. and Davis, J. N. 2012. LA Sprouts: a garden-based nutrition intervention pilot program influences motivation and preferences for fruits and vegetables in Latino youth. Journal of the Academy of Nutrition and Dietetics 112(6) 913-20.

Gottlieb, R. 2006. Reinventing Los Angeles; Nature and Community in the Global City. The MIT Press. Cambridge, Massachusetts, USA.

Guitart, D., Pickering, C., and Byrne J. 2012. Past results and future directions in urban community gardens research. Urban Forestry and Urban Greening 11:364-373.

Hale, J., Knapp, C., Bardwell, L., Buchenau, M., Marshall, J., Sancar, F, and Litt, J. S. 2011. Connecting food environments and health through the relational nature of aesthetics: Gaining insight through the community gardening experience. Social Science and Medicine 72, 1853-1863

Hanna, A. and Oh, P. 2000. Rethinking urban poverty: a look at community gardens. Bulletin of Science Technology and Society 20, 207–216.

Heraty, J.M. 2010. Conservation of maize germplasm as a result of food tradition in southern California's immigrant gardens. Masters Thesis. UC Davis.

Hooper, D. U., F. S. Chapin, J. J. Ewel, A. Hector, P. Inchausti, S. Lavorel, J. H. Lawton, D. M. Lodge, M. Loreau, S. Naeem, B. Schmid, H. Setala, A. J. Symstad, J. Vandermeer, and D. A. Wardle. 2005. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. Ecological Monographs 75:3-35.

Hope, D., Gries, C., Zhu, W. X., Fagan, W. F., Redman, C. L., Grimm, N. B., Nelson, A. L., Martin, C., and Kinzig A. 2003. Socioeconomics drive urban plant diversity. Proceedings of the National Academy of Sciences of the United States of America 100:8788-8792.

Huai, H., Xu, W., Wen, G., and Bai, W. 2011. Comparison of the Homegardens of Eight Cultural Groups in Jinping County, Southwest China. Economic Botany 65(4): 345–355.

Huang, G. L., W. Q. Zhou, and M. L. Cadenasso. 2011. Is everyone hot in the city? Spatial pattern of land surface temperatures, land cover and neighborhood socioeconomic characteristics in Baltimore, MD. Journal of Environmental Management 92:1753-1759.

Jackson, J., Rytel, K., Brookover, I., Efron, N., Hernandez, G., Johnson, E., Kim, Y., Lai, W., Navarro, M., Pena, A., Rehm, Z., Yoo, H., and Zabel, Z. 2013. Cultivate L.A.; An Assessment of Urban Agriculture in Los Angeles county. Public project prepared for the University of California Cooperative Extension, Los Angeles. Online available at: http://www.cultivatelosangeles.org

Jansson, A. and Polasky, S. 2010. Quantifying Biodiversity for Building Resilience for Food Security in Urban Landscapes: Getting Down to Business. Ecology and Society 15.

Jenerette G. D., Harlan, S. L., Stefanov, W., and Martin, C. 2011. Ecosystem services and urban heat riskscape moderation: Water, green spaces, and social inequality in Phoenix, USA. Ecological Applications 21:2637-2651

Kaplan, R., and Herbert, E.J. 1987. Cultural and sub-cultural comparisons in preferences for natural settings. Landscape and Urban Planning 14(4): 281-293.

Kinzig, A. P., Warren, P., Martin, C., Hope, D., and Katti, M. 2005. The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. Ecology and Society 10(1).

Koellner, T. and Schmitz, O. J. 2006. Biodiversity, ecosystem function, and investment risk. Bioscience 56:977-985.

Larsen, L., and Harlan, S. L. 2006. Desert dreamscapes: Residential landscape preference and behavior. Landscape and Urban Planning, 78(1/2), 85–100.

Lawson L. 2007. Cultural geographies in practice: The South Central Farm: Dilemmas in practicing the public. Cultural Geographies 14: 611-616.

Lawson, L. and Drake, L. 2013. Community Garden Organization Survey, 2011-2012. Community Greening Review. 18(20-41).

Lawton, J. H. 1999. Are there general laws in ecology? Oikos 84:177–192.

Liu, J. G., Dietz, T., Carpenter, S. R., Folke, C., Alberti, M., Redman, C. L., Schneider, S. H., Ostrom, E., Pell, A. N., Lubchenco, J., Taylor, W. W., Ouyang, Z. Y., Deadman, P., Kratz, T., and Provencher, W. 2007. Coupled human and natural systems. Ambio 36:639-649.

Loram, A., Thompson, K., Warren, P.H., and Gaston, K.J. 2008. Urban domestic gardens (XII): The richness and composition of the flora in five UK cities. Journal of Vegetation Science 19: 321–330.

Lovell, S. T. and Taylor J. R. 2013. Supplying urban ecosystem services through multifunctional green infrastructure in the United States. Landscape Ecology 28:1447-1463.

Lowry, J. H., Baker, M. E., and Ramsey, R. D. 2012. Determinants of urban tree canopy in residential neighborhoods: Household characteristics, urban form, and the geophysical landscape. Urban Ecosystems 15: 247–266.

Lubbe, C. S., Siebert, S. J., and Cilliers, S. S. 2011. Socio-economic drivers of plant diversity patterns in domestic gardens of the Tlokwe City Municipality. South African Journal of Botany **77**:538-539.

Manes, F., Incerti, G., Salvatori, E., Vitale, M., Ricotta, C., and Costanza, R. 2012. Urban ecosystem services: tree diversity and stability of tropospheric ozone removal. Ecological Applications 22:349-360.

Marco, A., Dutoit, T., Deschamps-Cottin, M., Mauffrey, J. F., Vennetier, M., and Bertaudiere-Montes, V. 2008. Gardens in urbanizing rural areas reveal an unexpected floral diversity related to housing density. Comptes Rendus Biologies 331:452-465.

Marco, A., Barthelemy, C., Dutoit, T., and Bertaudière-Montes, V. 2010. Bridging human and natural sciences for a better understanding of urban floral patterns: the role of planting practices in Mediterranean gardens. Ecology and Society 15(2): 2.

Mathieu, R., Freeman, C., and Aryal, J. 2007. Mapping private gardens in urban areas using object-oriented techniques and very high-resolution satellite imagery. Landscape and Urban Planning 81:179-192.

Matteson, K. C., Ascher, J. S., and Langellotto, G. A. 2008. Bee richness and abundance in New York city urban gardens. Annals of the Entomological Society of America 101:140-150.

Méndez, V.E., Lok., R. and Somarriba, E. 2001. Interdisciplinary analysis of homegardens in Nicaragua: micro-zonation, plant use and socio-economic importance. Agroforestry Systems 51:85-96.

Minkoff-Zern, L. A. 2012. Pushing the boundaries of indigeneity and agricultural knowledge: Oaxacan immigrant gardening in California. Agriculture and Human Values 29:381-392.

Peña, D. 2005. "Farmers Feeding Families: Agroecology in South Central Los Angeles" Lecture presented to the Environmental Science, Policy and Management Colloquium, October 10, 2005.

Peña, D. 2006. "Toward a critical political ecology of Latina/o urbanism". The Acequia Institute. Online available at: <http://www.acequiainstitute.org/researchreports.html>

Pickett, S.T.A., Cadenasso, M.L., Grove, J.M., Boone, C.G., Groffman, P.M., Irwin, E., Kaushal, S.S., Marshall, V., McGrath, B.P., Nilon, C.H., Pouyat, R.V., Szlavecz, K., Troy, A., and Warren, P. 2011. Urban ecological systems: Scientific foundations and a decade of progress. Journal of Environmental Management 92(3): 331-362.

Saldivar-Tanaka, L., and Krasny, M.E. 2004. Culturing community development, neighborhood open space, and civic agriculture: The case of Latino community gardens in New York City. Agriculture and Human Values 21(4): 399-412.

Schmelzkopf, K.1995. Urban community gardens as contested space. Geographical Review 85(3): 364-381.

Smith R.M., Gaston K.J., Warren P.H. and Thompson K. 2005. Urban domestic gardens (V): relationships between landcover composition, housing and landscape. Landscape Ecology 20: 235-253.

Smith, V. M., Greene, R. B., and Silbernagel. J. 2013. The social and spatial dynamics of community food production: a landscape approach to policy and program development. Landscape Ecology 28:1415-1426.

Soleri, D. and Cleveland, D.A. 2004. Farmer selection and conservation of crop varieties. In *Encyclopedia of plant and crop science*, (Goodman, R.M. ed.) 433-438. New York: Marcel Dekker.

U.S. Census Bureau: State and County QuickFacts: Los Angeles County. 2010. Available Online at: http://quickfacts.census.gov/qfd/states/06/06037.html

Van Heezik, Y., Freeman, C., Porter, S., and Dickinson K. J. M. 2013. Garden Size, Householder Knowledge, and Socio-Economic Status Influence Plant and Bird Diversity at the Scale of Individual Gardens. Ecosystems 16: 1442–1454.

Vavilov, N.I. 1949. The origin, variation, immunity, and breeding of cultivated plants. Chronica Botanica 13: 1-364

Wakefield, S., Yeudall, F., Taron, C., Reynolds, J., and Skinner, A. 2007. Growing urban health: Community gardening in South-East Toronto. Health Promotion International 22(2): 92-101.

Walker, J. S., Grimm, N. B., Briggs, J. M., Gries, C., and Dugan, L. 2009. Effects of urbanization on plant species diversity in central Arizona. Frontiers in Ecology and the Environment 7:465-470.

Wu, J. 2013. Landscape sustainability science: ecosystem services and human well-being in changing landscapes. Landscape Ecology 28(6):999–1023

Tables

Table 2.1: Descriptive statistics for all gardens, including tested factors of management style, ethnicity, garden age, median family income, and area of gardens and plots. Where number of gardeners exceeded number of plots, it meant that gardeners subdivided their plots with others or shared the work with family members. †There were over 100 plots, only a subsample of 69 was sampled.

Garden	Management	Ethnicity	Year founded	Income	Garden area (m^2)	Plot area (m^2)	Plots	Gardeners
IMM1	Individual	Asian	1988	$30,558	1440	46.46	32	32
IMM2	Individual	Hispanic	1999	$30,558	672	4.5	19	16
IMM3	Individual	Hispanic	2007	$49,006	4500	11.88	60	75
IMM4	Individual	Hispanic	1999	$29,927	819	9	26	25
IMM5	Individual	Hispanic	1989	$26,757	852	5.7	34	27
IMM6	Farm	Hispanic	1994	$25,161	9520	58.34	118	150
IMM7	Farm	Hispanic	1979	$53,150	2006	37	44	40
IMM8	Farm	Hispanic	2006	$25,161	23070	135	69†	69†
NIMM1	Individual	Non-Imm	2004	$82,676	10117	60	57	133
NIMM2	Individual	Non-Imm	2009	$45,478	930	7	32	32
NIMM3	Individual	Non-Imm	1989	$29,904	900	4.5	24	11
NIMM4	Individual	Non-Imm	1963	$70,774	448	17.5	16	16
NIMM5	Farm	Non-Imm	1996	$89,946	2244	52.63	25	20
NIMM6	Farm	African-American	1965	$25,161	6120	85	44	60

Table 2.2: Pearson's product moment correlation for hypothesized biodiversity mechanisms. Comparisons labeled (ALL) are for all gardens, while comparisons labeled (IND) are only for individually-based gardens.

	Income	Size	Age
Income (ALL)	–	0.04	.0.091
Size (ALL)	0.04	–	0.482**
Age (ALL)	.0.091	0.482**	–
Income (IND)	–	0.703*	0.322
Size (IND)	0.703*	–	0.794**
Age (IND)	0.322	0.794**	–

* P<0.05
** P<0.01

Table 2.3: Descriptive biodiversity across garden immigrant status and management styles. Garden indicates individually-based gardens and farm indicates communally-based. # of species is the number found in plots. Includes overall garden (n) and γ biodiversity for all species and each major species use

	Total	Immigrant Garden	Immigrant Farm	Non-immigrant Garden	Non-immigrant Farm
# of gardens	14	6	3	4	2
# of species	707	299	197	349	238
Edibles	229	160	135	152	105
Medicinals	44	26	19	27	16
Ornamentals	442	124	47	189	128

Table 2.4: Average Jaccard's dissimilarity index, divided into use categories, between plots in each specific garden (representative of plot turnover and β diversity). The higher the index, the more dissimilar garden plots are within that use. Different letters represent significant differences (P<0.01) between use types in a single garden. For all gardens, ornamental species were the most dissimilar within each garden.

		Species use†		
Garden ID		All	Edible	Ornamental
IMM1		0.862^{A}	0.873^{B}	0.985^{C}
	SE	0.002	0.002	0.002
IMM2		0.865^{A}	0.851^{B}	0.939^{C}
	SE	0.006	0.006	0.010
IMM3		0.874^{A}	0.863^{B}	0.994^{C}
	SE	0.002	0.002	0.001
IMM4		0.909^{A}	0.903^{A}	0.941^{B}
	SE	0.002	0.003	0.004
IMM5		0.916^{A}	0.908^{B}	0.977^{C}
	SE	0.002	0.002	0.003
IMM6		0.845^{A}	0.828^{B}	0.990^{C}
	SE	0.001	0.001	0.001
IMM7		0.867^{A}	0.858^{B}	0.940^{C}
	SE	0.002	0.002	0.004
IMM8		0.864^{A}	0.850^{B}	0.993^{C}
	SE	0.002	0.003	0.001
NIMM1		0.871^{A}	0.850^{B}	0.943^{C}
	SE	0.001	0.001	0.001
NIMM2		0.869^{A}	0.859^{B}	0.950^{C}
	SE	0.002	0.003	0.004
NIMM3		0.928^{A}	0.921^{A}	0.987^{B}
	SE	0.004	0.004	0.003
NIMM4		0.889^{A}	0.855^{B}	0.966^{C}
	SE	0.005	0.006	0.004
NIMM5		0.930^{A}	0.938^{A}	0.985^{B}
	SE	0.003	0.003	0.002
NIMM6		0.824^{A}	0.818^{A}	1.000^{B}
	SE	0.003	0.003	0.000

†Different letters represent significant differences (P<0.01) in Jaccard's index between use categories in each garden

Figures

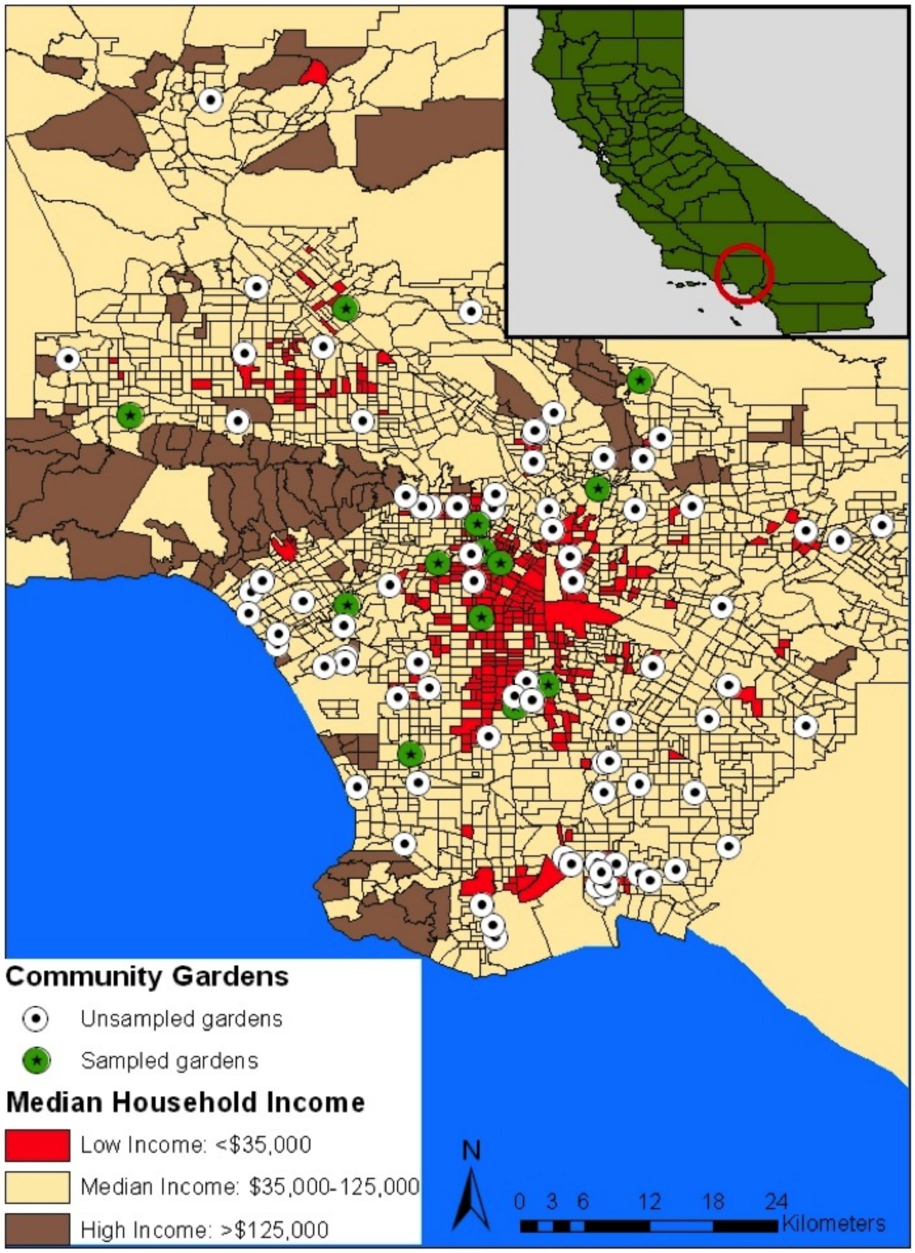

Figure 2.1: Map of Los Angeles County showing census tract boundaries (grey lines) and median household income variation (Red is low income, brown is high income, and beige indicates moderate income). The dots indicate the location of 99 community gardens in Los Angeles County, and the green points indicate surveyed locations.

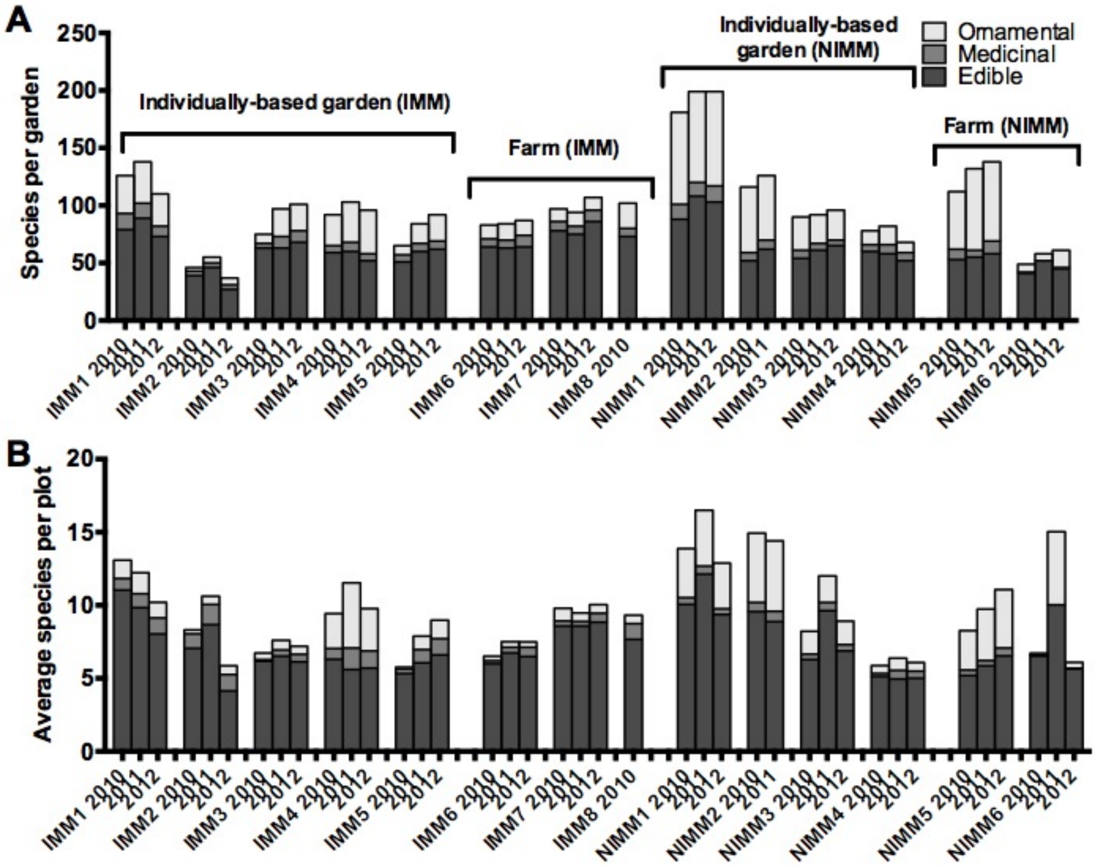

Figure 2.2: Descriptive garden scale (A) and plot scale (B) plant biodiversity according to major use categories (ornamental, medicinal, edible). Error bars in B indicate standard error for overall biodiversity of plots within a single garden. For both garden and plot biodiversity, a t-test indicated edible species in each garden were more bio-diverse than ornamental or medicinal species (Garden: P<0.001; Plot: P<0.001).

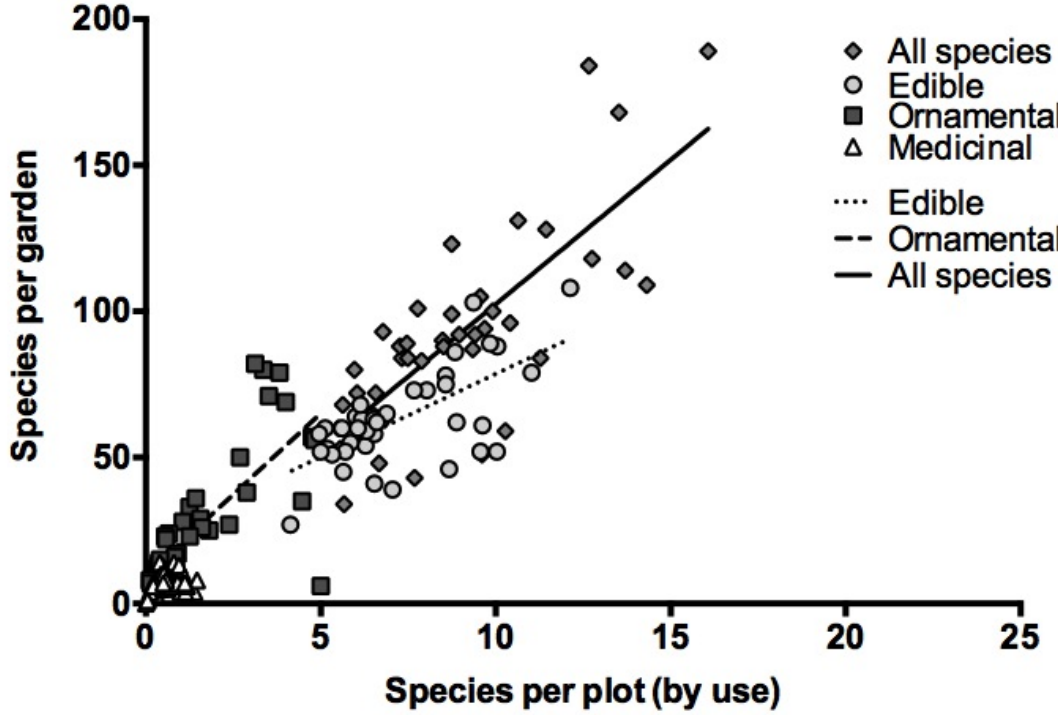

Figure 2.3: Relationship of species per garden (γ) to species per plot (α), divided into All species (diamonds; r^2=0.578 , p<0.001), Edible (circles; r^2=0.339, p<0.001), Medicinal (triangles; non-significant), and Ornamental species (squares; r^2=0.279 , p<0.001). γ diversity of each use is compared to average α diversity of each use. Each point represents a single community garden in a single year (~3 points per garden).

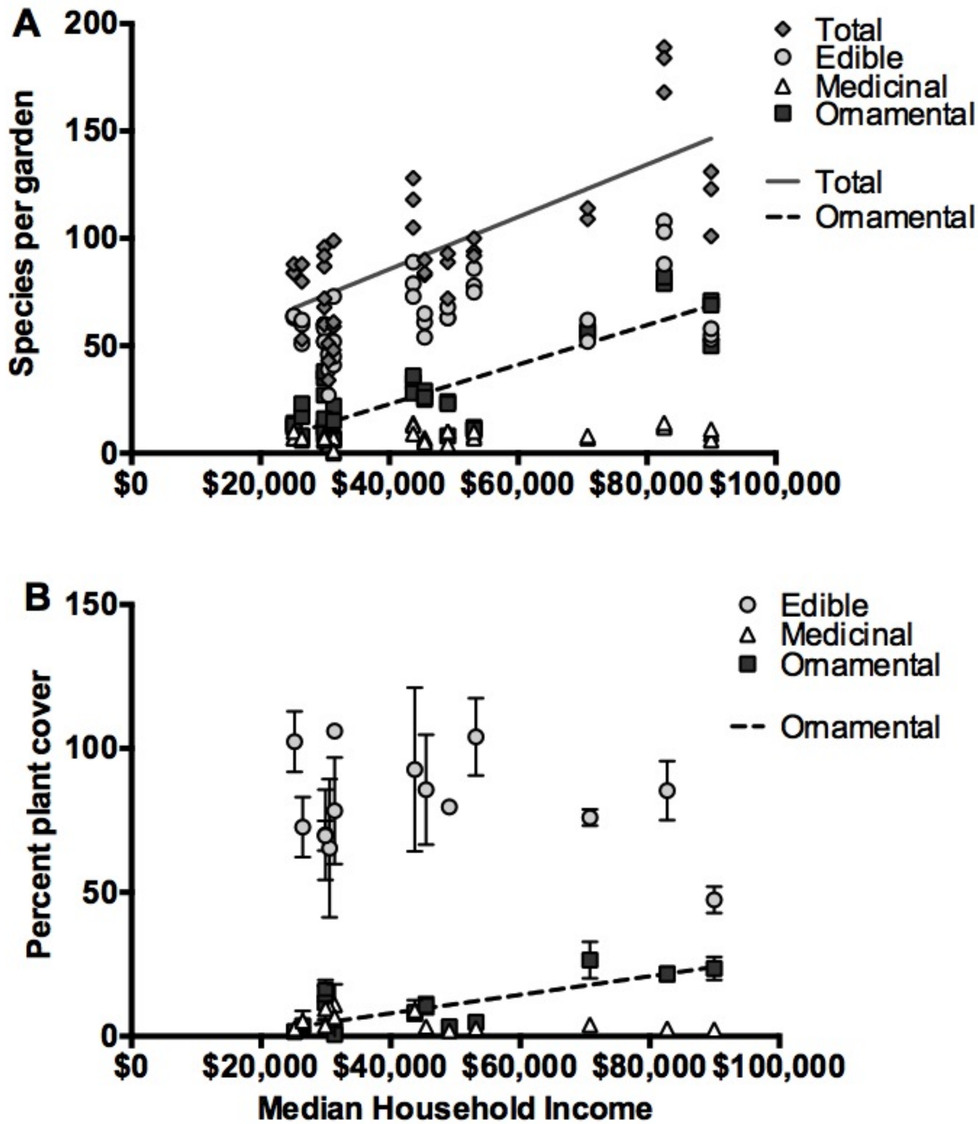

Figure 2.4: Relationships between neighborhood median income and biodiversity (A) and vegetation cover (B) for each of the major species uses. All species= diamonds, Edible=circles, Medicinal= triangles, and Ornamental= squares. Error bars represent variation between three survey years. All regressions reported are controlled for effect of plot size. Neighborhood income was related to total (r^2:0.468; $p<0.001$) and ornamental biodiversity ($r^2=0.620$;$p<0.001$) and to ornamental abundance (r^2: 0.530; $p<0.001$). Edible and medicinal richness and cover were not related to income.

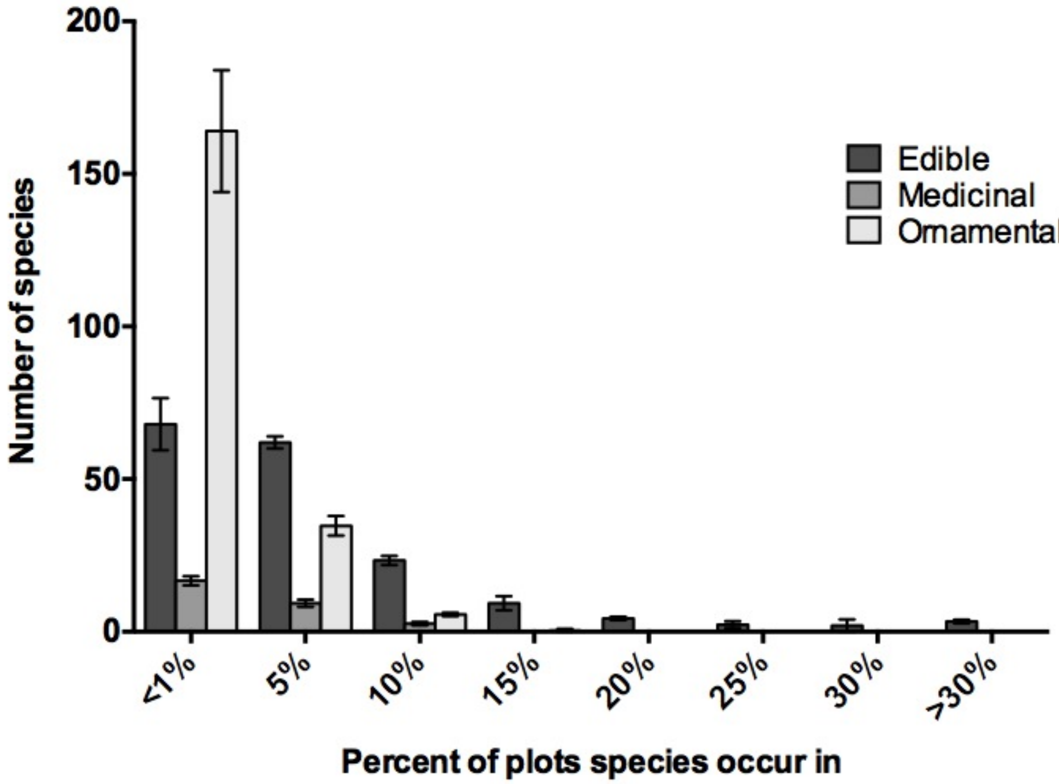

Figure 2.5: Frequency distribution of edible, medicinal, and ornamental species. The X-axis represents the percentage of plots across all gardens that contain a specific species and the Y-axis indicates how many species are present at that frequency. Error bars represents standard deviation between the three study years. No ornamental species were found in more than 10% of plots and most were found in less than 1% of plots. In contrast, most edibles were found in 10-30% of all plots.

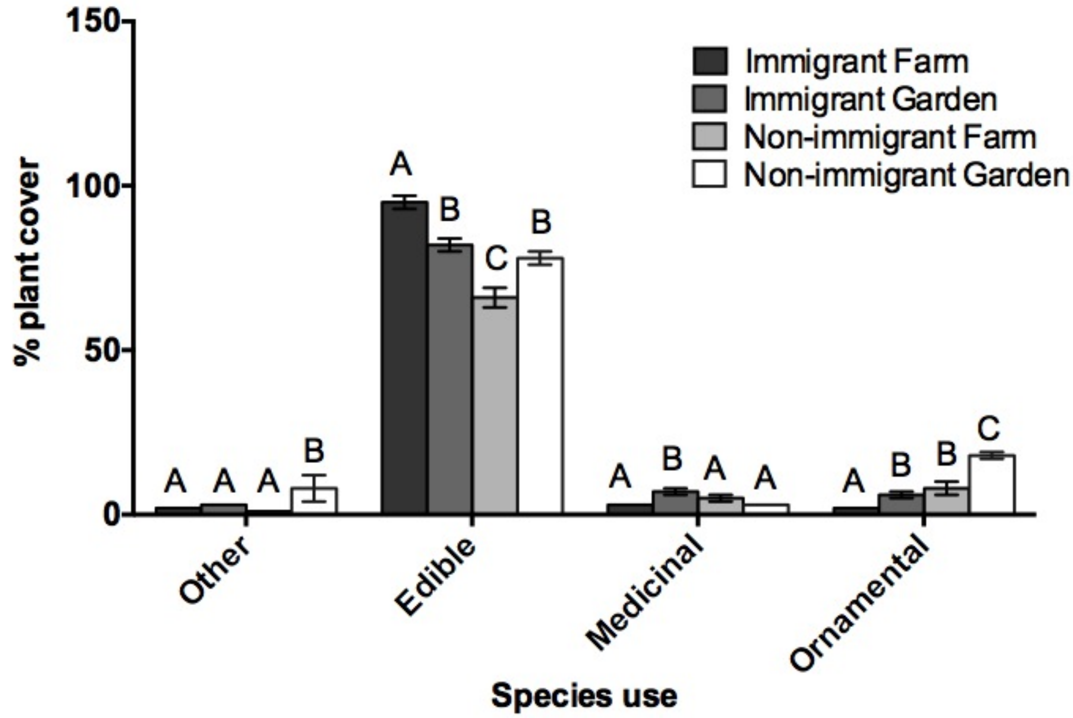

Figure 2.6: Average vegetative cover of species across uses, immigrant status, and garden management style. Error bars represent standard error across plots in specific garden categories in all three years. Different letters represent significant differences between cover of a specific use between garden categories.

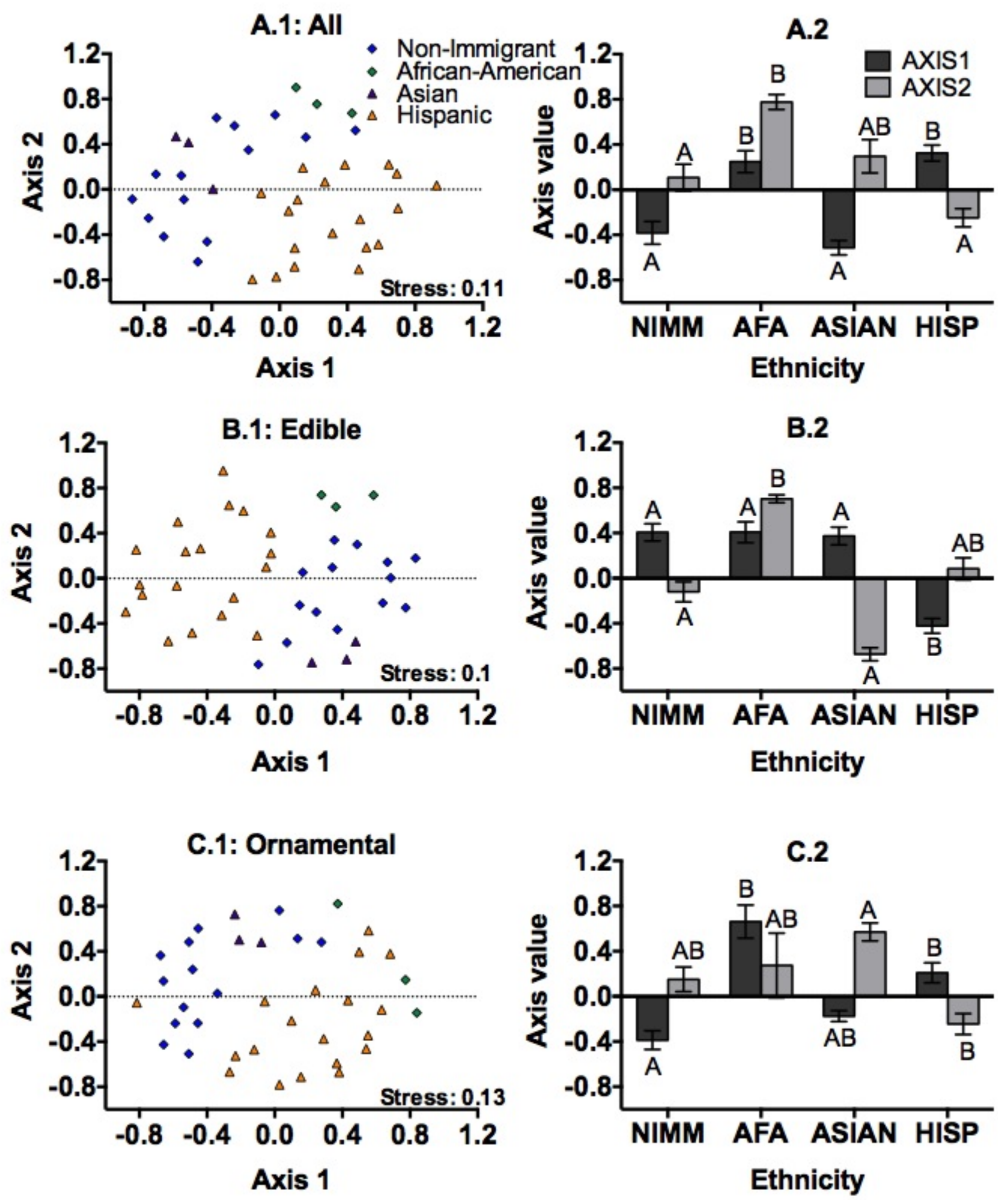

Figure 2.7: Column 1: NMDS (non-metric multidimensional scaling) ordination based on Jaccard's dissimilarity matrices for all species (A.1), Edible species (B.1), and Ornamental species (C.1). Each point represents a single garden in a single year. Stress levels in each plot indicate proportion of variance unaccounted for. Column 2: ANOVA comparing location of culturally distinct gardens on each ordination axis. Different letters indicate significant differences ($p<0.05$) between gardens of different ethnicities (AFA=African-American, ASIAN=Asian, HISP=Hispanic, NIMM=Non-immigrant) on that axis. Error bars represent standard error.

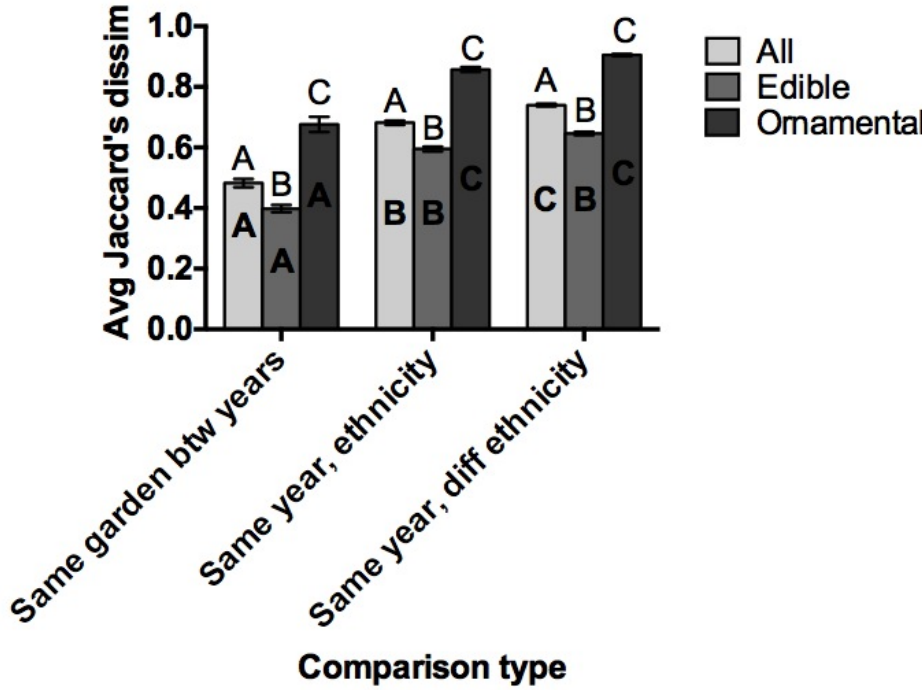

Figure 2.8: Average Jaccard's dissimilarity between gardens for major species uses (All, Edible, Ornamental). Comparisons include a single garden across each of three years, gardens in the same year and ethnicity, and gardens in the same year with different ethnicities. Different bold letters within columns represent significant differences between Jaccard's dissimilarity in a single use across comparison types. Different letters above columns represent significant differences between uses in a single comparison type. Error bars indicate standard error.

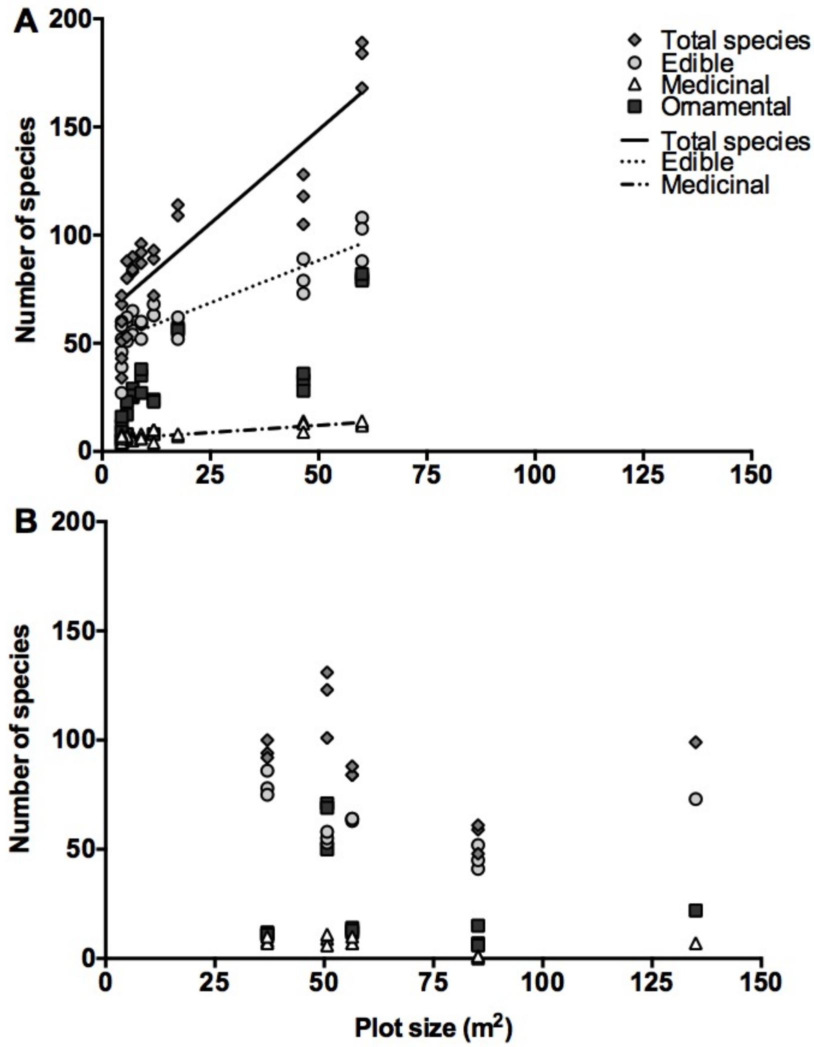

Figure 2.9: Relationship between plot size and species richness in individually based gardens (A) and farms (B). Total number of species (diamonds) is then divided into Edible (circles), Medicinal (triangles), and Ornamental (square) species. Each point represents one garden in one year, repeated for each use. Regression lines are based on analyses controlling for the effect of income. Plot size in individually based gardens (A) is positively related to all species (r^2=0.214; p<0.01), edibles (r^2=0.221; p<0.01), and medicinals (r^2=0.231; p<0.01), but not ornamentals (r^2=0.043; p=0.159). Plot size and biodiversity were not related in farms (B).

Appendix 1.1

Identity of most common Edible, Medicinal, Ornamental, and Other species. The column, # of occurrences, indicates the number of times each species occurred across all gardens in each year. If a species occurred in a single garden in 2010, 2011, and 2012, then # of occurrences would be 3, while if it occurred in three gardens in all three years, then it would be 9. Common names in italics are ethnic names for crops.

		Edible					
Common Name	**Scientific Name**	**Other**	**Edible**	**Medicinal**	**Ornamental**	**Family**	**# of Occurrences**
Zucchini	*Cucurbita pepo*		X			CUC	39
Tomato	*Solanum lycopersicum*		X			SOL	39
Onion	*Allium cepa*		X			ALL	38
Pepper	*Capsicum annum*		X			SOL	38
Cucumber	*Cucumis sativus*		X			CUC	38
Strawberries	*Fragaria ananassa*		X			ROS	37
Bean (Common)	*Phaseolus vulgaris*		X			FAB	37
Corn	*Zea mays*		X			POA	37
Spearamint	*Mentha spicata*		X			LAM	36
Chard	*Beta vulgaris ssp. cicla*		X			CHE	34
Beets	*Beta vulgaris ssp. vulgaris*		X			CHE	34
Broccoli	*Brassica oleracea Italica*		X			BRA	34
Garlic	*Allium sativum*		X			ALL	33
Serrano pepper	*Capsicum frutescens*		X			SOL	32
Basil	*Ocimum basilicum*		X			LAM	32
Giant Sunflower	*Helianthus annulus*		X		X	AST	32
Carrot	*Daucus carota*		X			API	31
Celery	*Apium graveolens*		X			API	30
Eggplant	*Solanum melongena*		X			SOL	30
Epazote	*Dysphania ambrosioides*		X	X		AMA	30
Cabbage	*Brassica oleracea Capitata*		X			BRA	29
Tomatillo	*Physalis philadelphica*		X			SOL	29
Radish	*Raphinus sativus*		X			BRA	29
Chives	*Allium schoenoprasum*		X			ALL	28

Kale	*Brassica oleracea Acephala*		X			BRA	28
Yam	*Ipomoea batata*		X			CON	28
Lettuce	*Lactuca sativa*		X			AST	28
Oregano	*Origanum vulgare*		X			LAM	28
Yerba Mora	*Solanum americanum*		X			SOL	28
Cilantro	*Coriandrum sativum*		X			API	27
Rosemary	*Rosmarinus officinalis*		X			LAM	26
Potato	*Solanum tuberosum*		X			SOL	26
Okra	*Abelmoschus esculentus*		X			Mal	24
Amaranthus	*Amaranthus hybridus*		X			AMA	24
Chayote	*Sechium edule*		X			CUC	24
Lemon Grass	*Cymbopogon citratus*		X			POA	23
French thyme	*Thymus vulgaris*		X			LAM	23
Shiso	*Perilla frutescens*		X	X		LAM	23
Watermelon	*Citrullis lantanus*		X			CUC	22
Chipilin	*Crotalaria longirostrata*		X			FAB	22
Butternut Squash	*Cucurbita moschata*		X			CUC	22
Chocolate Mint	*Mentha piperita*		X			LAM	22
Turnip Greens	*Brassica rapa*		X			BRA	21
Sugar Cane	*Saccharum sp.*		X			POA	21
Grapes	*Vitis vinifera*		X			VIT	21
Huauzontle	*Chenopodium nuttalliae*		X			CHE	20
Cantaloupe	*Cucumis melo*		X			CUC	20
Artichoke	*Cynara cardunculus*		X			AST	20
Parsley	*Petroselinum crispum*		X			API	20
Black eyed peas	*Vigna unguiculata*		X			FAB	20

Medicinal

Common Name	Scientific Name	Other	Edible	Medicinal	Ornamental	Family	# of Occurrences
Aloe Vera	*Aloe vera*			X		ASP	31
Epazote	*Dysphania ambrosioides*		X	X		AMA	30
Fever Few	*Chrysanthemum parthenium*			X	X	AST	28
Shiso	*Perilla frutescens*		X	X		LAM	23
Flor de Muerto	*Tagetes erecta*	Spiritual		X	X	AST	21
Ruta	*Ruta chalepensis*	Spiritual		X		RUT	18

Chamomile	*Matricaria recutita*			X		AST	12
Comfrey	*Symphytum officinale*			X		BOR	12
Kŭguā	*Momordica charantia*		X	X		CUC	10
Borage	*Borago officionalis*		X	X	X	BOR	10
Estafiate	*Artemisia ludoviciana*			X		AST	8
Yarrow	*Achillea millefolium*			X	X	AST	8
Acetillo	*Bidens pilosa*			X		AST	7
Minari	*Oenanthe javanica*		X	X		API	6
Mugwort (Common)	*Artemisia vulgaris*			X		AST	6
Rue, Common	*Ruta graveolens*			X		RUT	6
Plantain, common	*Plantago major*		X	X		PLA	5
Coneflower	*Echinacea sp.*			X	X	AST	5
Horsetail	*Equisetum sp.*			X	X	EQU	5
Loofah	*Luffa acutangula*	Hygiene	X	X		CUC	4
Castor Oil Tree	*Ricinus communis*	Shade		X		EUP	4
Coneflower	*Echinacea purpurea*			X	X	AST	4
Loofah	*Luffa cylindrica*	Hygiene	X	X		CUC	3
Jamaica	*Hibiscus sabdariffa*		X	X		Mal	3
Pericón	*Tagetes lucida*		X	X		AST	3
Dōngháncài	*Malva verticillata*			X		MAL	3
Shiso variety	*Perilla sp.*		X	X		LAM	2
Anise	*Pimpinella anisum*		X	X		API	2
Ajenjo	*Artemisia absinthium*			X		AST	2
White Ginger Lily	*Hedychium coronarium*			X	X	ZIN	2
Costmary	*Tanacetum balsamita*			X	X	AST	2
Tansy	*Tanacetum vulgare*			X	X	AST	2
Chaste Berry	*Vitex agnus-castus*			X	X	LAM	2
Cape gooseberry	*Physalis peruviana*		X	X		SOL	1
Agastache	*Agastache sp.*			X	X	LAM	1
Muicle	*Justicia spicigera*			X	X	LAM	1

<u>Ornamental</u>

Common Name	Scientific Name	Other	Edible	Medicinal	Ornamental	Family	# of Occurrences
Giant Sunflower	*Helianthus annulus*		X		X	AST	32
Fever Few	*Chrysanthemum parthenium*			X	X	AST	28

Roses	*Rosa sp.*				X	ROS	23
Flor de Muerto	*Tagetes erecta*	Spiritual		X	X	AST	21
Four O-clock Flower	*Mirabilis jalapa*				X	NYC	21
Zinnia	*Zinnia elegans*				X	AST	21
Gladiolas	*Gladiolus sp.*				X	IRI	19
Calla Lily	*Zantedeschia aethiopica*				X	ARA	19
Marigold	*Tagetes patula*	Pest Control			X	AST	18
Nasturtium	*Tropaeolum majus*	Shade	X		X	TRO	17
Garden Sage	*Salvia officinalis*		X		X	LAM	17
Morning Glory	*Ipomoea sp.*				X	CON	17
Pot Marigold	*Calendula officinalis*	Pest Control			X	AST	16
Chrysanthemum	*Chrysanthemum hybrid*				X	AST	14
Spanish Lavender	*Lavandula stoeches*				X	LAM	14
Garden Geranium	*Pelargonium hotorum*				X	GER	14
Mexican aster	*Cosmo bipinnatus*				X	AST	13
Lace Fern	*Asparagus setaceus*				X	ASP	12
California Poppy	*Eschscholzia californica*				X	PAP	12
Lemon Geranium	*Pelargonium crispum*				X	GER	11
Borage	*Borago officionalis*		X	X	X	BOR	10
Lablab	*Lablab purpureus*		X		X	FAB	10
Canna lily	*Canna generalis*				X	ZIN	10
Dahlia	*Dahlia sp.*				X	AST	10
Florist Kalanchoe	*Kalanchoe blossfeldiana*				X	CRA	10
Canna	*Canna hybrid*				X	ZIN	9
Amaryllis	*Hippeastrum sp.*				X	AMAR	9
Yarrow	*Achillea millefolium*			X	X	AST	8
Clavel de muerto	*Tithonia rotundifolia*	Spiritual			X	AST	8
Aeonium	*Aeonium sp*				X	CRA	8
Hollyhock	*Alcea rosea*				X	Mal	8
Peruvian Lily	*Alstroemeria sp. (hybrid)*				X	Lil	8
Rocket larkspur	*Delphinium ajacis*				X	RAN	8
Foxglove	*Digitalis sp.*				X	PLA	8
Bearded iris	*Iris sp. (subgenus iris)*				X	IRI	8

French Lavender	*Lavandula dentata*				X	LAM	8
African Daisy	*Osteospermum sp.*				X	AST	8
Sage, Mealycup	*Salvia farinacea*				X	LAM	8
Shasta Daisy	*Chrysanthemum maximum*				X	AST	7
Carnation	*Dianthus carophyllus*				X	CARO	7
Gerbera Daisy	*Gerbera hybrida*				X	AST	7
Ghost plant	*Graptopetalum paraguayense*				X	CRA	7
Chinese hibiscus	*Hibiscus rosa sinensis*				X	Mal	7
Kalanchoe	*Kalanchoe sp.*				X	CRA	7
Woolflower	*Celosia plumosa*				X	AMA	6
Batchelor Buttons	*Centaurea cyanus*				X	AST	6
Spider plant	*Chlorophytum comosum*				X	AGA	6
Spotted Snapweed	*Impatiens balsamina*				X	BAL	6
Lily	*Lilium sp.*				X	LIL	6
Shanin	*Petunia violacea*				X	SOL	6
Salvia hybrid	*Salvia sp.*				X	LAM	6
Pansies	*Viola sp.*				X	VIO	6

<u>Other</u>

Common Name	**Scientific Name**	**Other**	**Edible**	**Medicinal**	**Ornamental**	**Family**	**# of Occurrences**
Century Plant	*Agave americana*	Fiber			X	AGA	5
Saltwort	*Salsola sp.*	Fiber				AMA	2
Ivy (common)	*Hedera helix*	Ground cover			X	ARA	3
Loofah	*Luffa acutangula*	Hygiene	X	X		CUC	4
Vietnamese Gourd	*Luffa cylindrica*	Hygiene	X	X		CUC	3
Marigold	*Tagetes patula*	Pest Control			X	AST	18
Pot Marigold	*Calendula officinalis*	Pest Control			X	AST	16
Gopher Perch	*Euphorbia lathyris*	Pest control				EUP	10
Nasturtium	*Tropaeolum majus*	Shade	X		X	TRO	17
European Nettle Tree	*Celtis australis*	Shade				CAN	4
Castor Oil Plant	*Ricinus communis*	Shade		X		EUP	4

Japanese Pine	*Pinus thunbergiana*	Shade			X	PIN	4
White Mulberry	*Morus alba*	Shade	X			MOR	3
Tree of Heaven	*Alianthus altissima*	Shade				SIM	3
Tipuana	*Tipuana tipu*	Shade				FAB	3
Elm	*Ulmus sp.*	Shade				ULM	3
Mexican fan palm	*Washingtonia robusta*	Shade			X	ARE	3
Golden Rain	*Koelreuteria paniculata*	Shade				SAP	2
American Elm	*Ulmus americana*	Shade				ULM	2
Yucca	*Yucca sp.*	Shade			X	AGA	2
Flor de Muerto	*Tagetes erecta*	Spiritual		X	X	AST	21
Rue, Fringed	*Ruta chalepensis*	Spiritual		X		RUT	18
Clavel de muerto	*Tithonia rotundifolia*	Spiritual			X	AST	8

Chapter 3: Drivers of plant biodiversity and ecosystem service production in home gardens across the Beijing Municipality of China

Abstract:

Home gardens have been recognized as repositories of agricultural biodiversity across the world. The influence of socioeconomics and location-specific factors on urban gardening patterns merits continued study. Using Beijing Municipal Province in China, a rapidly urbanizing region, as a case study, I address two questions: 1) How do biodiversity patterns change between different urbanized regions in Beijing? 2) How do ecosystem services provided by Beijing home gardens change with socioeconomic status and location-based preferences of gardeners? I surveyed 104 home gardens in Beijing Municipal Province for plant biodiversity, abundance, and species ecosystem services (ES) (provisioning or cultural uses). The gardens were distributed across three urbanized regions (suburban, peri-urban, and exurban). I found that species biodiversity and abundance shift according to a hierarchy of need from ornamentals (cultural ES) to edibles (provisioning ES) with increasing distance from Beijing. These trends are related to reduced income, lowered food security, and lack of urban markets in exurban regions. Rarefaction curves indicate ornamental species drive diversity. Ordination also showed a shift in species composition with increasing isolation from the city; Suburban and exurban gardens were the most different, while peri-urban gardens were similar to both others. Only exurban gardens had a positive relationship between species and area. High edible cover and high species density indicates that demand for edibles in exurban

regions may be higher than space constraints allows. This study better quantifies species biodiversity patterns in Beijing, and can inform urban planners about the value and usefulness of home garden space.

Introduction

Home gardens are potentially hotspots of agricultural biodiversity in urban regions (Arriess and Clawson 1994; Nguyen 2003; Kumar and Nair 2004; Galluzzi et al. 2010), which stand in contrast to mono-cultured commercial croplands. They are a ubiquitous landscape across the world, with an estimated 15-36% of residential land in the UK, India, Africa, and China occupied by home gardens (Loram et al. 2008; Davies et al. 2009; Cilliers et al. 2012; Huai et al. 2011; Jaganmohan et al. 2012; Baudry and Yu 1999). Variation in garden biodiversity and abundance can be high, even within a single urbanized region, due to socioeconomic or cultural status of residents (Lubbe et al. 2011; Cilliers et al. 2012; Jaganmohan et al. 2012). Research on home gardens outside of Europe has been primarily focused on rural gardens (Del Angel-Pérez and Mendoza 2009; Huai et al. 2011), though some recent work has examined urban regions in the developing world (Molebatsi et al. 2010; Lubbe et al. 2011; Jaganmohan et al. 2012). Home gardens have been shown to maintain local food security (Wezel and Bender 2003), especially in the rapidly urbanizing regions of the developing world (Cilliers et al. 2013). My study aims to quantify the biodiversity and ecosystem services produced in home garden ecosystems in villages across an urbanized gradient in Beijing, China. Quantitative studies of home garden agro-ecosystems can provide opportunities for rapid increases in fundamental knowledge of how biological organization directly affects local nutrition, biodiversity, and global food security.

Beijing, China is one of the most rapidly urbanizing regions in the world, and its food systems are threatened with rapid and extensive conversion of agriculture to urban and non-agricultural uses (Ho and Lin 2004; Zhang et al. 2006). China must feed 22% of the world's population on 6.4% of the global land area, 7.2% of the world's farmland, and 5.8% of the world's annual water resource (CCICCD 1996). The structure and size of agricultural land in China has been changing since reforms in the 1970s (Baudry and Yu 1999). The current challenge for home gardens outside of Beijing is their uncertain land tenure; land use policy enacted in 1995 states that agricultural land around the city cannot be effectively protected by the government unless it is competitive with other urban land uses (Zhang et al. 2009). Quantifying the value of home gardens as compared to urban developments and improved transportation connectivity can be difficult, especially in areas with reduced socioeconomic resources.

This study incorporates an ecosystem service (ES) approach to home garden research in Beijing, China, focusing on how demand for certain services, and thus plant choice changes spatially with the needs of residents across an urbanizing gradient (McDonnell and Hahs 2009; Cilliers et al. 2013). In particular, I focus on provisioning and cultural services, both shown to be valued in home garden systems (Galluzzi et al. 2012; Lubbe et al. 2010). Chinese urban and rural settlements have a long cultural history of home gardens, having both provisioning and cultural ES for participants (Baudry and Yu 1999; Huai and Hamilton 2009; Huai et al. 2011). Increased biodiversity in landscapes can also provide indirect supporting ES such as soil nutrient cycling, pollinator biodiversity, and biological control of pests. For example, Beijing villages with biodiverse field margins and home gardens have higher carabid biodiversity, important predators for agricultural systems (Yu et al. 2006). High biodiversity in edible, ornamental, and shade plants contribute to provisioning and cultural ES production, as well as supporting ES (Galluzzi et al. 2010; Mitchell and Hanstad 2004).

Socioeconomic factors have been widely shown to influence plant biodiversity in human dominated ecosystems (Hope et al. 2003; Kinzig et al. 2005). One framework to better understand regional socioeconomic effects on garden species choice is a hierarchy of needs (Lubbe et al. 2011; Clarke et al. 2013). Within this framework gardeners are expected to plant species according to their needs, from food and medicine to aesthetics. Rural villages generally have reduced local income and access to urban food markets as compared to urban dwellers (Zimmer and Kwong 2004) and may be expected to select garden species with provisioning ES, like edibles or medicinals with less emphasis on aesthetic species (Lubbe et al. 2010; Cilliers et al. 2012). In addition, agricultural knowledge and participation has been closely linked with edible biodiversity in rural areas; this secondary hypothesis indicates that gardeners who rely monetarily on garden success may be more likely to plant edible species (Fu et al. 2006; Lubbe et al. 2011; Galluzzi et al. 2012). In contrast, higher incomes and access to urban markets in suburban villages may cause a garden composition shift towards ornamentals, which provide aesthetic and cultural ES. A pattern of increased ornamental diversity and decreased edible abundance correlated with income has also been observed across Europe (Galluzzi et al. 2010; Loram et al. 2008).

Local agricultural traditions and preferences may also influence composition of crops providing a specific ES, resulting in less species turnover between gardens in a single urban region (Barau et al. 2013). Participant agricultural background and local traditions have been closely linked to preferences for specific edible crops in urban agricultural spaces, suggesting that villages who share agricultural experience will also share species compositions (Fu et al. 2006; Lubbe et al. 2011; Galluzzi et al. 2012). Reduced road access and distance from local markets can further influence biodiversity patterns by reducing the need to grow cash crops, which can create unique patterns of species in rural villages (Abebe et al. 2013). Peri-urban villages, intermediate between rural and suburban villages, combine agricultural participation with intermediate income and market access and may contain gardens with both high edible and ornamental biodiversity which overlap compositionally with both suburban and rural gardens.

As resident needs change across a distance and socioeconomic gradient from city boundaries, so may garden management and species density. Variation in species-area relationships, the change in number of species with habitat area (Koellner and Schmitz 2006), is often indicative of community assembly processes in natural and human dominated ecosystems (Gotelli and Colwell 2001; Breuste et al. 2008). Studies in both European and Asian home gardens indicate a positive linear relationship between garden size and species biodiversity (Smith et al. 2005; Loram et al. 2008; Huai et al. 2011; Abebe et al. 2013). Unmet demand for provisioning ES may drive strong relationships between species and garden area within a specific village. Variation in species-area relationships between villages may result from differences in agricultural knowledge across urbanized regions. Increased agricultural knowledge and greater plant needs in exurban villages may encourage complex garden structures in more rural regions where garden sizes are constrained (Arriess and Clawson 1994; Kumar and Nair 2004). In contrast suburban villages may have a more limited palate of species they can cultivate and may plant a similar number of species independent of available garden space. Such patterns of size-invariant species planting has been shown in some French home gardens in densely population regions (Marco et al. 2008).

This multi-scale agricultural study describes the vegetative composition and ES produced in home gardens in five villages in three urbanized regions, suburban, peri-urban, and exurban, within the Beijing Municipality of China. These regions are organized along a distance gradient from the city, as isolation from urban resources impact income, population density, access to urban markets, and occupation; a pattern observed in cities generally and specifically in Beijing (McDonnell and Hahs 2008; Huai et al. 2011; Yunlai and Fengying 2009; Table 3.1). I focus on how the coupling between socioeconomic status, access to markets, and agricultural knowledge of residents in each region affects overall biodiversity and species uses (whether plants provide provisioning or cultural/aesthetic ES). Studying socioeconomic change across an urbanizing gradient provides a framework for incorporating residents into ecological system dynamics (Alberti et al. 2003; McDonnell et al. 2012; Boone et al. 2012).

My study focuses on answering two distinct questions: 1) *How do biodiversity patterns change between different urbanized regions in Beijing?* 2) *How do ecosystem services provided by Beijing home gardens change with socioeconomic status and location-based preferences of gardeners?* In answering these questions I address hypotheses that describe the selection and biodiversity across gardens in different urbanized regions and species uses within each garden. My analyses provide data for comparisons with home gardens across the world to help quantify their overall contribution to urban biodiversity and ES.

Methods

Study Area

The Beijing municipality on the northeast coast of China spans 16,800 sq. km, with a population of over 20 million people, with a 54% increase since 2001 (National Bureau of Statistics 2010; Beijing Bureau of Statistics 2012). Of these, 86% of total residents reside in urbanized Beijing, and over 35% of the total population includes migrants from other provinces (Beijing Bureau of Statistics 2012). Residents living in exurban areas in China earn less than half as much as their urban counterparts (Zimmer and Kwong 2004), and are more likely to get their income from farm activities (CCICCD 1996; Huai and Hamilton 2009). Beijing municipality contains some of the country's most productive agricultural land. The shape and management of agricultural land near Beijing has changed since agricultural reforms were instituted in 1978 (Yu et al .1999). These reforms de-collectivized state land and instituted an individual household-based farming system (Lin 1992). Enactment of these reforms has encouraged more vegetable farming, and expanded both home gardens and multi-crop productivity throughout China (Yu et al.1999; Baudry and Yu 1999; Ouyang et al. 2004). Agricultural land is in decline, with the loss of over 545,000 ha of agricultural land near Beijing to urbanization in the last 20 years (Ho and Lin 2004). Urban growth policies in Beijing province include the replacement of courtyard centered villages with more compact modern housing, reducing local planting space in the process (Drew 2008; Kessell and Gillet 2011).

Data Collection

Five different villages across a distance gradient from the border of Beijing, China were sampled for home garden biodiversity and ES production. These villages were located in one of three different urbanized regions, defined by their distance from the fifth ring road (the city border) in Beijing (Figure 3.1). For each region, villages representative of regional environmental and socioeconomic variability were chosen, based on population density, number of households, income, agricultural production, and elevation. I determined population density through the National Bureau of Statistics (NBS), while percent of people in agricultural jobs, and village area were established through visits to local government offices (Table 3.1). Income per capita was estimated through combined NBS estimates and interviews with village officials. Even so, many forms of income go unreported; NBS income data does not include income generated from household property (e.g. rentals) or unofficial income from crops sold from farmland (Sicular et al. 2007). Supplemental income was aggregated from previous interviews with local gardeners and officials, though village scale variation may be larger than the sample. Despite these limitations, these income estimates correspond well to other urban to rural estimates in China (Zimmer and Kwong 2004; Yunlai and Fengying 2009; Huai et al. 2011), and I believe they are appropriate for my analysis scale.

The closest village to Beijing, ShangZhuang (SHZ), was located less than 15 km from the city border, and was the only suburban village. Though relatively small in total area, SHZ is home to ~3500 individuals and ~2000 households and is typical of suburban development around Beijing (Table 3.1). Agricultural production is lowest in this village. Two villages, DongXinZhuang (DXZ) and XinZhouYing (XZY), were sampled in the peri-urban region, ~30 km from the city border. A higher percentage of residents from these two villages work in agriculture than in suburban villages and more of the village area is dedicated to agricultural use. At around 55 km from Beijing, near the base of the Yanshan mountains, my research group visited two exurban villages, NiuPenYu (NPY) and XiaoShuiYu (XSY) (Figure 3.1). These villages were large (9 and 19 km^2) and contained the lowest density of households. Near 40% of villagers work in agricultural jobs, and both income per capita and cost of living are low.

To representatively sample gardens in each village, stratified random sampling procedures were followed to include a statistically robust number of gardens (Bartlett et al. 2001). The total number of households in each village was obtained through national census and local government offices. My research group conducted focused interviews with city officials to estimate the number of households containing home gardens within each village. The goal was to sample 5-10% of the existing gardens within each village.

On average, 42% of households have home gardens. With a total of 2,985 households across all five villages (Table 3.1), it was estimated that 1,254 contained gardens. According to a statistical method for estimating adequate sample size in a given population (outlined in Bartlett et al. 2001), sample size should be 108 total gardens (3% margin of error, a=0.05). As each village varied in number of households, we visited between 15-30 gardens per village, for a total combined sample of 104 gardens, close to the target sample goal.

As villages do not keep accurate home garden census records, my group visited home gardens opportunistically. Using maps of the villages, we visited each of the four quadrants of the village over the course of a few days. We walked the length of streets in that region and looked for residents at home. If a garden was present in that household courtyard, we asked for permission to survey their garden, regardless of crop coverage. Residents were open to the survey, and my research group was only denied entry twice. We then asked each participant about households with active gardens in this village quadrant, effectively identifying new survey participants (a technique outlined in Russell, 2006). For two of the villages, XZY and XSY, we were able to work more closely with village officials, who set up appointments with 5-10 households with gardeners. Remaining samples were identified through interviews with these participants.

For each visited home garden, I estimated garden (delineated region used for planting) and parcel size (space owned by residents, including courtyard, shed, and house). As many dwellings had complex structures, gardens and parcels were difficult to measure directly. Instead, smaller identifiable structures were measured for reference (e.g. a 30 m^2 shed) and then full size was estimated by sight and interview with household members. All deliberately cultivated plants and trees were identified and percent cover of each species estimated. Larger tree canopies were measured on-site and smaller plant cover percentages were estimated visually. Species, not subspecies or specific varieties, were recorded, with a few exceptions. If different parts of the plant were used between varieties or the local use was different, I recorded them separately. For instance, some *Cucurbita pepo* subspecies were used as decorative gourds, and defined separately from food species. Residents were asked about the identity of any unknown species. Proper taxonomic identification for any unusual species was assured through photos and collection of voucher specimens for expert identification and archiving at the China Agricultural University herbarium. In addition, gardeners were asked about how each species was used, determining if each species provided provisioning or aesthetic/cultural ES. Use categories included edibles (E) and medicinals (M), both provisioning uses, and ornamentals (O), plants with cultural or aesthetic service value. In addition, I include an "Other" category (D) for less common provisioning services, which included shade, timber, fiber, fencing, or windbreaks. Many species had multiple uses and were noted once for each use, thus making the accumulated number of all individually used species greater than overall diversity.

Analysis

To compare garden biodiversity and abundance between villages, I conducted analyses of variance (ANOVA) at two size scales, garden and village (SPSS 16.0). Plot level biodiversity and abundance were compared between individual villages and between urbanization regions. These were further separated into number and abundance of each species use (edible, medicinal, ornamental, and other). Size of gardens and parcels was compared similarly at the village and region scale. In order to evaluate compositional differences between villages and urbanization regions, I conducted a principal components ordination using a program previously developed by Exequiel Ezcurra and used for biodiversity assessments (Altesor et al. 1998; Garcillán and Ezcurra 2003). The program solves both Correspondence Analysis (Hill 1973) and Principal Component Analysis (Noy-Meir 1973, 1975) as an eigenvector decomposition problem. The eigensolutions are calculated using the numerical algorithm proposed by Press et al. (2007). Garden similarity was compared within calculated 2D ordination space and individual species were projected onto the same ordination space.

I conducted a linear regression to assess the relationship of cover and biodiversity values to size of the garden plot and parcel (SPSS 16.0). These regressions were repeated for each garden, urbanization region, and all plant uses within them. Since parcel and plot size are intrinsically linked (a garden plot is limited by the overall size of the parcel), I first conducted a controlled regression to determine which had the most influence over biodiversity and cover. This controlled regression indicated that, while garden size is significantly correlated to parcel size, all observed relationships between species-area and cover-area were only significant for garden size. Therefore, I report species-area relationships for garden plot, not parcel size.

I constructed sample-based rarefaction curves, randomized and smoothed species accumulation curves, for each sampled region to compare α diversity (regional biodiversity), sampling adequacy, and species saturation. Rarefaction curves are produced by repeatedly re-sampling the pool of N samples, so measures of α biodiversity can be directly compared at any sampling intensity (Gotelli and Colwell 2001; Colwell et al. 2004). While I am confident that this sampling strategy has produced a representative sample of all possible garden-containing households (Bartlett et al. 2001), some garden species are likely missing. One difficulty in sampling managed vegetation is that even exhaustive sampling may not produce an asymptotic curve, indicating sufficient sampling effort. In addition, a rigorous comparison of rarefaction curves also requires well-defined confidence intervals, which, until recently, were less reliable, as they were based on sample size (Colwell et al. 2012). I use a new technique to extrapolate rarefaction curves (described in Colwell et al. 2012; EstimateS 9.0), which resamples observed data stochastically. This technique is more robust than analytical models, allowing estimates of the number of plots needed to reach asymptote, biodiversity at asymptote, and confidence intervals independent of original sample size. I extrapolated rarefaction curves to 90 gardens for each region to equalize regions and for comparison to other studies, which often include 100 or more gardens per urbanized region (Jaganmohan et al. 2012; Lubbe et al. 2011; Huai et al. 2011). I also calculated a species richness estimate, using the first order jackknife estimator, which minimizes bias and allows estimation of total species without an asymptotic species accumulation curve. This estimator is a function of rare species; with every rare (n) species found, the jackknife estimate is 1/n(n-1) more than the total number found (Heltshe and Forrester 1983), and is calculated based on observed samples. As I hypothesize species composition will be different across regions, rarefaction and extrapolation are shown separately for all species, edible species, and ornamental species at the regional scale.

β diversity, turnover between gardens, can also be estimated regionally using rarefaction curves modeled with a power law function – $y=Cx^z$ – where C is a constant and z is the slope of the function (Koellner et al. 2004). The exponent, z, is a measure of β diversity in each region, as it describes the rate of species accumulation (Arita and Rodriguez 2002; Zhao et al. 2010; Clarke et al. 2013). The slope of z ranges from 0-1, with 1 indicating that there are no shared species between gardens in a given region (high β diversity) and 0 indicating identical species in each garden in a given region (low β diversity). Power law functions were based on the extrapolated 90 garden sample created by EstimateS, and repeated for each region and plant use.

Results:

Suburban gardeners have the highest income per capita and fewest jobs in agriculture, indicating financial and physical access to city resources, while exurban gardeners have the highest agricultural participation and lowest income per capita (Table 3.1). Village population and density is highest near the city and decreases towards exurban areas. Finally, climatic variables of average temperature and precipitation both decrease with distance from the city. Though a few very large gardens were found in exurban areas (>500 m^2), garden size generally ranged between 150-200 m^2. The similarity of sizes was supported by an ANOVA showing no significant differences in garden area between villages or urbanization regions ($p>0.05$).

Overall, 278 distinct species and sub-species were found across the five villages, most of which were in the edible (100) or ornamental (152) use category (Table 3.2). Individual villages had between 76 and 163 species across all gardens (Table 3.2). Exurban villages had similar numbers of edible species as peri-urban (70 and 79 respectively), despite having fewer sampled gardens (29 vs. 45 gardens). Suburban and peri-urban gardens contained more unique ornamentals found solely in that region (30 and 41 respectively) compared with exurban villages (7). While differences between medicinal and other species uses were not clear at the village scale, peri-urban gardens had the highest biodiversity of those uses as well.

Near 50% of all ornamental species and 30% of edible species were unique to one of the 5 villages. Diversity differences were more pronounced at the regional level, and peri-urban gardens had a higher number of species than suburban or peri-urban ($p<0.05$; Figure 3.2b). Peri-urban gardens have high ornamental biodiversity similar to suburban gardens, while retaining high edible biodiversity, resulting in the highest garden scale biodiversity ($p<0.05$; Figure 3.2a,b). For cover, no individual villages had significant differences between any use category of cover (Figure 3.3a). When gardens were grouped into regions, edible cover was highest in exurban villages (Figure 3.3b).

Over all 104 gardens, the species accumulation curve did not reach asymptote at the observed 278 species or extrapolated to 200 gardens, indicating species will further increase if more gardens were added (Figure 3.4). This appeared to be due to the steady increase of ornamental species, which were estimated to increase from 152-181 species with the addition of 100 gardens (Figure 3.4). In contrast, edible species diversity was near asymptote at 104 gardens, with only 10 more species extrapolated for the addition of over 100 more gardens. Likewise, species accumulation curves did not reach asymptote individually in any of the three urbanized regions based on the 30-45 gardens sampled, (Figure 3.5a). Extrapolated continuations of the rarefaction curves showed that at near 90 gardens per urbanized region, suburban and exurban gardens were nearing asymptote, at an estimated 240 and 159 species respectively (Figure 3.5a; Table 3.3), while peri-urban regions were still gaining species at 244 species. These numbers are supported by the first order jackknife indicator, which calculated that asymptote would be reached for suburban gardens at 236 (+/-16) species and exurban at 165 (+/-11), while peri-urban gardens would not reach asymptote until 270 (+/- 9) species (Table 3.3). Confidence intervals for all species overlapped for suburban and peri-urban regions, indicating they did not have significantly different numbers of species; however, both regions had significantly more species than exurban gardens with non-overlapping confidence intervals (Figure 3.5a). For edible species, all three urbanized regions had overlapping accumulation curve confidence intervals (Figure 3.5b), indicating that each region was not significantly different in species richness, even though the per-garden comparisons show suburban gardens with fewer edible species (Figure 3.2b). All three regions reach edible biodiversity saturation at 90 gardens, results supported by the first order jackknife indicator, as confidence intervals overlap with the calculated asymptotic range (Table 3.3). For ornamental species, suburban and peri-urban accumulation curves overlap in confidence intervals, indicating that they have similar ornamental composition and exurban garden ornamental biodiversity is much lower (Figure 3.5c), data supported by per garden comparisons (Figure 3.2c). Extrapolation to 90 gardens does not show ornamental saturation in any region, (Figure 3.5c) supporting the unique ornamental composition in each region (Table 3.2).

Suburban gardens had the highest overall beta (β) diversity (0.59) in comparison to peri-urban (0.47) and exurban (0.48) regions, indicating greater species turnover between suburban gardens (Table 3.3). β diversity was lower for edible species in peri-urban (0.34) and exurban regions (0.37), though notably less so between suburban gardens (0.46). In contrast, ornamental β diversity was the highest among all uses, indicating decreased species overlap in garden ornamentals in all regions (suburban: 0.67; peri-urban: 0.54; exurban: 0.64).

Peri-urban gardens showed compositional similarity with both suburban and exurban areas (Figure 3.6), supporting diversity similarities observed for both suburban ornamentals and exurban edibles (Figure 3.2). Ordination also indicated that gardens in exurban villages have the most similar species compositions to each other, as indicated by their similar location on the ordination space (Figure 3.6). An ANOVA showed no difference between exurban gardens and XZY on axis 1, while DXZ and SHZ gardens occurred in different locations (Figure 3.6). Along axis 2, the two peri-urban villages had no significant differences, while the cluster of XSY, an exurban village, was in a significantly different area than all others. Because of the high number of species found in only one or two gardens across an entire village, variation between gardens was too high to distinguish species groupings.

Garden and parcel sizes were not distinctly different between villages or urbanization regions. Across all village types, only exurban garden diversity was related to size (r^2=0.440, p<0.001; Figure 3.7). When broken down into use types, edible (r^2=0.337, p<0.001), medicinal (r^2=0.196, p=0.016), and ornamental (r^2=0.154, p=0.034;) species all increased with garden size in exurban villages.

Discussion

The results of this intensive study provide comprehensive information on home garden biodiversity and species uses and their regional variation near a megacity of China. Such information is currently limited for cities in developing countries, although essential for the quantification of ES and human well being in locations of rapid urbanization (Cilliers et al. 2013; Jaganmohan et al. 2012; Lubbe et al. 2011). These data show high α and β diversity across all villages, with distinct species composition for each urbanized region (Figure 3.4; 3.5; Table 3.3). Quantitative data on biodiversity and species cover across multiple urbanized regions may aid in local protection for agricultural land by making it competitive with other urban land uses (Zhang et al. 2009).

Hierarchy of need

One important result of my study is that species uses and ES production in home gardens change across an urbanizing gradient, supporting a hierarchy of need hypothesis. Poorer exurban communities with less access to urban markets are more likely to select garden species providing edible, medicinal, shade, and other provisioning ES than more affluent suburban and peri-urban communities (Figure 3.2b; Figure 3.5). These plants may provide additional income or improve gardener livelihoods through providing a basic need (Lubbe et al. 2010; Cilliers et al. 2012; Cilliers et al. 2013). The higher number of ornamental species and decreased edible cover in suburban and peri-urban gardens (Figure 3.2, 3.3) may be attributed to luxury investment in cultural and ornamental ES as well as decreased provisioning needs. This shift from cultural to provisioning services with distance from the city and declining socioeconomic status has been observed in home gardens across the world (Thaman et al. 2006; Bernholt et al. 2009; Lubbe et al. 2010; Cilliers et al. 2012).

Biodiversity patterns

Purposeful plant biodiversity was extensive in home gardens, with 278 species found across the three urbanized regions and 337 estimated in the extrapolated species accumulation curve of 200 gardens (Table 3.2; Table 3.3). Peri-urban and suburban regions had not reached asymptote at the study sampling intensity, and were only approaching when extrapolated to 90 gardens (Table 3.3; Figure 3.5a), suggesting that species will further increase with more gardens. More intensive studies than ours, with 100-300 gardens sampled per urbanization region, also did not reach species saturation (Lubbe et al. 2011; Cilliers et al. 2012; Jaganmohan et al. 2012). Species saturation is unlikely in managed garden systems, as species choices are only limited by the available plant pool at nurseries, which can range into thousands of species (Smith et al. 2005).

The main driving force behind the non-saturating species accumulation curves appears to be unique ornamental species; of the 152 ornamental species found in the survey, 78 of them were unique to a single region and most of those were only found in a single garden (Table 3.2). This is also reflected in the high β diversity of ornamental species in peri-urban and suburban regions (Table 3.3; Figure 3.5c). Indeed, the only urbanized region estimated to reach species saturation at 90 gardens was in exurban villages (Figure 3.5a), which have the highest edible coverage and biodiversity and are significantly lower in ornamental biodiversity (Figure 3.2b; 3.3b). Since every region reached saturation in edible species (Figure 3.5b), low β diversity within exurban gardens is unsurprising (Table 3.3). Likely, the need for provisioning ES in exurban villages translates to a reduced demand for cultural ES such as aesthetics, which drive α and β biodiversity (Cilliers et al. 2013). Though overall garden β diversity (0.45-0.56) is relatively high in comparison to temperate deciduous forests (0.2-0.4; Connor et al. 1983; Koellner et al. 2004), some highly urbanized areas have even greater heterogeneity (0.7-0.9; Clarke et al. 2013). I interpret the intermediate β diversity as a product of ES demand; though high ornamental novelty is desired, especially near the city (β=0.54-0.67; Qian et al. 2007), provisioning species are more constant, with lower turnover within a region (β=0.34-0.45), tempering garden species turnover.

Though my sample size of 104 gardens was a relatively small subset of total village area, (<20,000 m^2 total), an extensive biodiversity survey within Beijing found only 500 weedy and cultivated species in over 42,800 m^2 in the urban landscape (Wang et al. 2012) as compared to my 278. Overall, this dense biodiversity in home gardens is consistent with research from other cities showing that home garden biodiversity surpasses most other urban land uses (Lubbe et al. 2010; Lubbe et al. 2011). The biodiversity level I found is comparable to a study of 300 home gardens in rural India (n=258; Jaganmohan et al. 2012), and 100 low income urban gardens in Africa (n=270; Cilliers et al. 2013), and is three times more diverse than a recent survey of 15 villages in Southwestern China (Huai et al. 2011). In contrast, peri-urban and urban regions of home gardens in other developing countries may be much more extensive, as shown by a recent 100 garden survey in South Africa with over 800 cultivated species (Lubbe et al. 2011). Considering the high β diversity for ornamental species in this survey, regional valuation of cultural services and differences in socioeconomic status between my study region and those in past studies may account for the disparity in observed biodiversity patterns (Bernholt et al. 2009; Cilliers et al. 2012).

Local traditions and regional species composition

While food species diversities across urbanized regions overlap heavily in species accumulation curves (Figure 3.5b), ordination shows strong differences between suburban and exurban village species compositions, with peri-urban villages similar to both suburban and exurban villages (Figure 3.6). I suggest these changes occur from linked local traditions and economic factors, as agricultural participation, ES demand, and socioeconomic factors all varied by both village and urbanized region. Similar compositional species shift have also been observed in African home gardens, where the main food species grown in gardens changes from leafy vegetables near the city to grain crops in deep rural villages (Molebatsi et al. 2010). Isolation from major markets may have further encouraged distinct species biodiversity in exurban regions (Abebe et al. 2013). As a large disparity exists between food security in urban vs. exurban areas across China (Yunlai and Fengying 2009), exurban gardeners may cultivate crops uncommon to commercial farms, due to scarcity of specialized provisioning species outside of urban markets in China (Qian et al. 2007; Akinnfesi et al. 2010).

Garden size-species relationships

A relationship between individual home garden size and number of species has been observed in multiple countries, including other villages in China (Loram et al. 2008; Kabir and Webb 2009; Huai et al. 2011). Surprisingly, I only found a relationship between garden size and species diversity in exurban villages (Figure 3.7). Reduced income in exurban regions coupled with reduced availability of food markets contributes to lowered food security, making productive gardens necessary to local food systems (Kabir and Webb 2009; Yunlai and Fengying 2009; Galluzzi et al. 2010). In addition, specific food needs fulfilled by each edible species cannot be substituted by replacement with other species (Peña 2006). A complex garden structure with multiple plant layers is usually observed in more rural regions (Michon and Mary 1994; Del Angel-Perez and Mendoza 2004; Akkinfesi et al. 2010), a pattern reflected in this study where exurban gardens had edible species cover of over 100% (Figure 3.3). The high edible cover and complex vertical garden structure indicates that demand for species diversity in exurban regions may be higher than space constraints allows. With increased space, more species are planted to address demand, leading to the observed species-area relationship. For peri-urban and suburban gardens, species are not planted as intensely, indicating that space does not limit species choice. If a gardener desires ten food species and all ten species can be grown in the available space, increased increments of garden space may be used for expanding existing species, not adding new varieties. Other studies that show a consistent species-area relationship across all urbanized regions indicate that garden space is often insufficient to support all local species needs (Kabir and Webb 2009; Loram et al. 2008; Albuquerque et al. 2005). Gardeners in exurban regions also have been shown to have a higher agricultural knowledge base than their urban counterparts, and are therefore more able to maintain a maximum species density in their gardens (Thaman et al. 2006; Airriess and Clawson 1994; Albuquerque et al. 2005). Therefore, exclusively exurban species-area relationships can be explained through a combination of the space-species demand mismatch and agricultural ability in exurban areas to maintain high species densities.

Conclusions and Implications

My research provides quantitative data on biodiversity, species abundance, and the ways participants use gardens to supplement their health and well-being. The mechanisms regulating garden biodiversity that I propose in this paper (hierarchy of need, local agricultural traditions, size of managed area) can be applied broadly to urban garden systems across the world, an essential part of advancing urban ecological science

(McDonnell and Hahs 2013). I show that urban garden biodiversity shifts across different urbanized regions in Beijing according to a hierarchy of need. Gardeners change from cultivating aesthetically pleasing species (cultural ES) to more useful edible species (provisioning ES) with increasing isolation from the city and decreased socioeconomic status. Edible and ornamental composition also shifted, possibly due to cultural shifts between suburban and exurban villages. Surprisingly, I also show that the hierarchy of needs also influences species area relationships; low-income exurban communities may have greater demand for species than they have planting space, leading to a clear increase in species with any new increment of space.

Large-scale agriculture has outcompeted many small farmers in China, but lack local vegetable varieties (Yunlai and Fengying 2009), and many varieties of crops in China are vanishing due to reduced traditional ecological knowledge (Pei et al. 2010; Huai and Hamilton 2009). Though I did not identify individual varieties of common vegetables, other studies show that home gardens can be germplasm banks for the conservation of local varieties, preserving agricultural biodiversity (Huai et al. 2011; Levasseur and Olivier 2000). Policies to encourage biodiversity in Chinese farmlands are sparse, and diversified cropping systems seen in home gardens are lacking in modern farms (Liu et al. 2011). Food production in Chinese gardens reduces the demand on commercial agriculture (Zhang et al. 2006) and may increase local food security (Wezel and Bender 2003; Huai et al. 2009). My study highlights how biodiversity in home gardens changes along socioeconomic gradients, shifting from cultural to provisioning ES with decreased gardener income and access to important food resources.

Works Cited:

Abebe, T., Sterck., F.J., Wiersum, K.F., and Bongers, F. 2013. Diversity, composition and density of trees and shrubs in agroforestry homegardens in Southern Ethiopia. Agroforestry Systems 87:1283–1293.

Aguílar-Stoen, M., Moe, S.R., and Lucia Camargo-Ricalde, S. 2009. Home Gardens Sustain Crop Diversity and Improve Farm Resilience in Candelaria Loxicha, Oaxaca, Mexico. Human Ecology 37(1): 55-77.

Airriess C. A., and Clawson D. L. 1994. Vietnamese market gardens in New Orleans. Geographical Review. 84(1):16-31.

Akkinfesi F.K., Sileshi G.W., Ajayi O.C., Akinnifesi, A.I., de Moura E.G., Linhares J.F.P., and Rodrigues, I. 2010. Biodiversity of the urban homegardens of São Luís city, Northeastern Brazil. Urban Ecosystems. 13: 129-146

Albuquerque, U.P., Andrade L.H.C., and Caballero, J. 2005. Structure and floristics of homegardens in Northeastern Brazil. Journal of Arid Environments 62: 491–506

Altesor, A., Di Landro, E., May, H. and Ezcurra, E. 1998. Long-term species change in a Uruguayan grassland. Journal of Vegetation Science 9(1): 173–180.

Alberti, M., Marzluff, J.M., Shulenberger, E., Bradley, G., Ryan, C., and Zumbrunnen, C. 2003. Integrating humans into ecology: Opportunities and challenges for studying urban ecosystems. Bioscience 53(12): 1169-1179.

Arita, H. T., and Rodríguez, P. 2002. Geographic range, turnover rate and the scaling of species diversity. Ecography 25: 541–550.

Bartlett, J. E., Kotrlik, J.W., and Higgins, C.C. 2001. Organizational Research: Determining Appropriate Sample Size in Survey Research. Information Technology, Learning, and Performance Journal 19(1): 43-50.

Baudry, J. and Yu, Z. 1999. Landscape patterns changes in two subtropical Chinese villages as related to farming policies. Critical review of Plant Science 18 (3): 373-380.

Barau, A.S., Ludin, A.N.M, and Said, I. 2013. Socio-ecological systems and biodiversity conservation in African city: Insights from Kano Emir's Palace gardens. Urban Ecosystems16:783–800.

BMBS (Beijing Municipal Bureau of Statistics) 2012. China Statistics Press, Beijing (in

Chinese).

Bernholt H., Kehlenbeck K., Gebauer J., and Buerkert A. 2009. Plant species richness and diversity in urban and peri-urban gardens of Niamey, Niger. Agroforestry Systems. 77: 159-179.

Boone, C. G., Cook, E., Hall, S. J., Nation, M. L., Grimm, N. B., Raish, C. B., Finch, D. M., and York, A. M. 2012. A comparative gradient approach as a tool for understanding and managing urban ecosystems. Urban Ecosystems 15:795-807.

Breuste, J., Niemela, J., and Snep, R. P. H. 2008. Applying landscape ecological principles in urban environments. Landscape Ecology 23:1139-1142.

China National Committee for Implementing the United Nations Convention to Combat Desertification (CCCICCD). 1996. China country paper to combat desertification, China Forestry Publishing House, Beijing, pp 18–31

Cilliers, S., Cilliers, J., Lubbe, R. and Siebert, S. 2013. Ecosystem services of urban green spaces in African countries—perspectives and challenges. Urban Ecosystems (September) 1-22.

Cilliers, S., Siebert, S., Davoren, E., and Lubbe, R. 2012. Social aspects of urban ecology in developing countries, with an emphasis on urban domestic gardens. *In: Applied Urban Ecology: A Global Framework*, First Edition. (Matthias Richter and Ulrike Weiland, eds). Blackwell Publishing Ltd.

Clarke, L. W., Jenerette G.D., and Davila, A. 2013. The luxury of vegetation and the legacy of tree biodiversity in Los Angeles, CA. Landscape and Urban Planning 116:48-59

Colwell, R. K., Mao, C. X., & Chang, J. 2004. Interpolating, extrapolating, and comparing incidence-based species accumulation curves. Ecology. 85(10): 2717–2727

Colwell, R. K., Chao, A., Gotelli, N. J., Lin, S.-Y, Mao, C. X., Chazdon, R. L., and Longino, J. T. 2012. Models and estimators linking individual-based and sample-based rarefaction, extrapolation, and comparison of assemblages. Journal of Plant Ecology 5:3-21.

Connor, E. F., McCoy, E. D., and Cosby, B. J. 1983. Model discrimination and expected slope values in species-area studies. The American Naturalist, 122, 789–798.

Davies, Z.G., Fuller, R.A., Loram, A., Irvine, K.N., Sims, V., and Gaston, K.J. 2009. A national scale inventory of resource provision for biodiversity within domestic gardens. Biological Conservation 142(4): 761-771.

Del Angel-Pérez, A.L. and Mendoza, B.M.A. 2004. Totonac homegardens and natural resources in Veracruz, Mexico. Agriculture and Human Values 21 (4): 329-346.

Drew, J. 2008. "In Beijing, No Answer to the Bulldozer". Washington Post, Saturday, April 2008. Online available at < http://www.washingtonpost.com/wp-dyn/content/article/2008/04/25/AR2008042503503.html>

Fu, Y., F., Guo, H., Chen, A., and Cui J. 2006. Household differentiation and on-farm conservation of biodiversity by indigenous households in Xishuangbanna, China. Biodiversity Conservation 15:2687–2703.

Galluzzi, G., Eyzaguirre, P., and Negri, V. 2010. Home gardens: neglected hotspots of agro-biodiversity and cultural diversity. Biodiversity and Conservation 19(13): 3635-3654.

Galluzzi G. 2012. 'Agrobiodiversity and protected areas: another approach to synergies between conservation and use?' In: On-farm conservation of neglected and underutilized species: status, trends and novel approaches to cope with climate change – Proceedings of the International Conference, Friedrichsdorf, Frankfurt, 14-16 June, 2011

Garcillán, P.P. and Ezcurra, E. 2003. Biogeographic regions and beta-diversity of woody dryland legumes in the Baja California peninsula. Journal of Vegetation Science 14(6): 859–868.

Gotelli, N. J. and Colwell, R. K. 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. Ecology Letters 4:379-391.

Heltshe, J. F. and Forrester, N. E. 1983. Estimating species richness using the jackknife procedure. Biometrics, 39(1): 1–11.

Hill, M. O. 1973. Reciprocal averaging: an eigenvector method of ordination. Journal of Ecology 61: 237-249.

Ho, S.P.S., and Lin, G.C.S. 2004. Non-agricultural land use in post-reform China. China Quarterly 179: 758-781.

Hope, D., Gries, C., Zhu, W. X., Fagan, W. F., Redman, C. L., Grimm, N. B., Nelson, A. L., Martin, C., and Kinzig A. 2003. Socioeconomics drive urban plant diversity. Proceedings of the National Academy of Sciences of the United States of America 100:8788-8792.

Huai, H. and Hamilton, A. 2009. Characteristics and functions of traditional homegardens: A review. Frontiers of Biology in China 47:151– 157.

Huai, H., Xu, W., Wen, G., and Bai, W. 2011. Comparison of the Homegardens of Eight Cultural Groups in Jinping County, Southwest China. Economic Botany 65(4): 345–355.

Jaganmohan, M., Vailshery, L. S., Gopal, D. and Nagendra, H. 2012. Plant diversity and distribution in urban domestic gardens and apartments in Bangalore, India. Urban Ecosystems 15: 911-925.

Kabir, M. E. and Webb, E.L. 2009. Household and homegarden characteristics in southwestern Bangladesh. Agroforestry Systems 75:129–145.

Kessel J. and Gillet K. 2011. "The Fate of Old Beijing: The Vanishing Hutongs" Online documentary, produced by China Green. Online available at: http://sites.asiasociety.org/chinagreen/feature-hutong/

Kinzig, A. P., Warren, P., Martin, C., Hope, D., and Katti, M. 2005. The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. Ecology and Society 10(1).

Kirkpatrick J.B., Daniels G.D., and Zagorski T. 2007. Explaining variation in front gardens between suburbs of Hobart, Tasmania, Australia. Landscape and Urban Planning 79: 314-322.

Koellner, T., Hersperger, A. M., and Wohlgemuth, T. 2004. Rarefaction method for assessing plant species diversity on a regional scale. Ecography, 27, 532–544.

Koellner, T. and Schmitz, O. J. 2006. Biodiversity, ecosystem function, and investment risk. Bioscience 56:977-985.

Kumar, B. M., and Nair, P. K. R. 2004. The Enigma of Tropical Homegardens. Agroforestry Systems 61: 135–152.

Levasseur V, and Olivier A. 2000. The farming system and traditional agroforestry systems in the Maya community of San Jose, Belize. Agroforestry Systems 49:275–288.

Lin J.Y. 1992. Rural reforms and agricultural growth in China. The American Economic Review 82 (1): 34-51.

Liu Y, Duan M, and Yu Z. 2011. Agricultural landscapes and biodiversity in China.

Agriculture, Ecosystems & Environment 166:46-54.

Loram, A., Thompson, K., Warren, P.H., and Gaston, K.J. 2008. Urban domestic gardens (XII): The richness and composition of the flora in five UK cities. Journal of Vegetation Science 19: 321–330.

Lubbe, C.S., Siebert, S.J., and Cilliers, S.S. 2010. Political legacy of South Africa affects the plant diversity patterns of urban domestic gardens along a socio-economic gradient. Scientific Research and Essays 5:2900–2910

Lubbe, C.S., Siebert, S.J., and Cilliers, S.S. 2011. Floristic analysis of domestic gardens in the Tlokwe City Municipality, South Africa. Bothalia 41(2): 351-361.

Marco, A., Dutoit, T., Deschamps-Cottin, M., Mauffrey, J.F., Vennetier, M., and Bertaudiere Montes, V. 2008. Gardens in urbanizing rural areas reveal an unexpected floral diversity related to housing density. Comptes Rendus Biologies 331(6): 452-465.

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

McDonnell, M.J., and Hahs, A.K. 2013. The future of urban biodiversity research: Moving beyond the **'**low-hanging fruit**'.** Urban Ecosystems 16:397–409.

McDonnell, M. J., Hahs, A. K., and Pickett, S. T. A. 2012. Exposing an urban ecology straw man: critique of Ramalho and Hobbs. Trends in Ecology & Evolution 27:255-256.

Michon G. and Mary F. 1994. Conversion of traditional village gardens and new economic strategies of rural households in the area of Bogor, Indonesia. Agroforestry Systems 25: 31–58.

Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: synthesis. Island press, WA, 155 pp.

Mitchell, R. and Hanstad, T. 2004. Small homegarden plots and sustainable livelihoods for the poor. LSP Working paper no.11. Food and agriculture organization of the United Nations, Rome, Italy

Molebatsi L.Y., Siebert S.J., Cilliers, S.S., Lubbe, C.S. and Davoren E. 2010. The Tswana tshimo: A homegarden system of useful plants with a particular layout and function. African Journal of Agricultural Research 5(21): 2952-2963

National Bureau of Statistics, China, 2010. Online available at: http://www.stats.gov.cn/english/statisticaldata/yearlydata/

Nguyen, M.L.T. 2003. Comparison of food plant knowledge between urban Vietnamese living in Vietnam and in Hawai'i. Economic Botany 57(4): 472-480.

Noy-Meir, I., 1973. Data transformation in ecological ordination: I. Some advantages of non-centering. Journal of Ecology. 61: 329- 341.

Noy-Meir, I., Walker, D. & Williams, W. T., 1975. Data transformation in ecological ordination: II. On the meaning of data standardization. Journal of Ecology 63: 779-800.

Ouyang, J., Song, C., Yu, Z., Zhang, F. 2004. Farm households choice of land use type and its effectiveness on land quality and environment in Huang-Huai-Hai Plain. Journal of Natural Resources 19(1): 1-11 (In Chinese)

Pei, S., Hamilton A.C., Yang L., Huai, Y., Yang, Z., Gao, F., and Zhang, Q. 2010. Conservation and development through medicinal plants: A case study from Ludian (Northwest Yunnan, China) and presentation of a general model. Biodiversity and Conservation 19 :2619–2636.

Peña, D. 2006. "Toward a critical political ecology of Latina/o urbanism". The Acequia Institute. Electronic document <http://www.acequiainstitute.org/researchreports.html>

Press, W.H., S.A. Teukolsky, W.T. Vetterling, & B.P. Flannery. 2007. Numerical Recipes 3rd Edition: The Art of Scientific Computing (3rd edition). Cambridge University Press, Cambridge, U.K.

Qian, T., Joyce, D., and He, S. 2007. China's ornamentals industry is in 'full bloom'. Centre for Native Floriculture Report. March, 2007. 6 pp.

Russell, B.H. 2006. *Research methods in anthropology: Qualitative and quantitative methods,* 2nd ed. Lanham, MD: Altamira Press.

Sicular T., Ximing Y., Gustafsson B., and Shi L. 2007. The Urban-Rural Gap and Income Inequality in China. Review of Income and Wealth 53 (1): 93-126

Smith R.M., Gaston K.J., Warren P.H., and Thompson K. 2005. Urban domestic gardens (V): relationships between landcover composition, housing and landscape. Landscape Ecology 20: 235-253.

Thaman, R.R., Elevitch, C.R., and Kennedy J. 2006. Urban and homegarden agroforestry in the Pacific Islands: Current status and future prospects. In: Kumar BM, Nair PKR (eds) *Tropical homegardens: a time-tested example of sustainable agroforestry. advances in agroforestry*, vol 3. Springer, Dordrecht, pp 25–41

Wang H., MacGregor-Fors, and López-Pujol, J. 2012. Warm-temperate, immense, and sprawling: plant diversity drivers in urban Beijing, China. Plant Ecology 213: 967-992.

Wezel A. and Bender S. 2003. Plant species diversity of homegardens of Cuba and its significance for household food supply. Agroforestry Systems 57: 37–47.

Yu, Z.R., Baudry, J., Zhao, B.P., Zhang, H., and Li, S.Q. 1999. Vegetation components of a subtropical rural landscape in China. Critical Reviews in Plant Sciences 18(3): 381-392.

Yu, Z.R., Liu, Y., and Axmacher, J.C. 2006. Field margins as rapidly evolving local diversity hotspots for ground beetles (Coleoptera : Carabidae) in northern China. Coleopterists Bulletin 60(2): 135-143.

Yunlai, X. and Fengying, N. 2009 A report on the status of China's food security. China Agricultural Science and Technology Press, Beijing, China.

Zhang, K., Li, X., Zhou, W., Zhang, D., and Yu, Z. 2006. Land resource degradation in China: Analysis of status, trends and strategy. International Journal of Sustainable Development and World Ecology 13(5) 397-408.

Zhang, F., Cai, J., and Liu, G. 2009. How urban agriculture is reshaping peri-urban Beijing? Open House International 34(2): 15-24.

Zhao, J., Ouyang, Z., Zheng, H., Zhou, W., Wang, X., Xu, W., and Ni, Y. 2010. Plant species composition in green spaces within the built-up areas of Beijing, China. Plant Ecology 209: 189-204.

Zimmer, Z. and Kwong, J. 2004. Socioeconomic Status and Health among Older Adults in Rural and Urban China. Journal of Aging and Health 16(1): 44-70.

Tables

Table 3.1: Description of village level characteristics for 5 sampled villages. Population density and income/capita were obtained through the National Bureau of Statistics, while village density, % of people in agricultural jobs, and village area were established through visits to local government offices.

Variable	Suburban	Peri-urban		Exurban	
	Urbanization region				
Village abbreviation	SHZ	DXZ	XZY	XSY	NPY
Income/capita (yuan)	13755	10172	11000	6000	5900
Distance from 5^{TH} ring road (km)	14	30	32	59	61
Average temp (°C)	12.1	11.5	11.5	10.8	10.8
Population density (person/km^2)	1262	381	455	61	32
Households in village	1917	294	350	214	210
Area of village (m^2)	2795.5	2574	1775	19000	8960
% residents with agricultural jobs	9.67%	28.5%	32.4%	40%	37.9%
Average garden size (m^2)	131	207	210	237	157
Gardens sampled	30	30	15	17	12

Table 3.2: Number of species in each sampled village and each urbanization region, divided into plant uses. Unique species refers to those found only in that village or region. The number of edible species was similar between peri-urban and exurban gardens, while the number of ornamentals was similar between peri-urban and suburban gardens.

Villages											
Use category	**SHZ**		**XZY**		**DXZ**		**NPY**		**XSY**		**All villages**
	Total	Unique	Total	Unique	Total	Unique	Total	Unique	Total	Unique	
Other	11	3	12	1	13	5	7	2	10	1	23
Edible	54	5	58	6	64	11	51	6	54	3	100
Medicinal	15	3	11	3	16	7	3	0	9	3	27
Ornamental	92	30	57	9	87	26	20	2	26	4	152
Total	157	44	119	17	163	45	76	10	84	13	278

Urbanization region							
Use category	**Suburban**		**Peri-Urban**		**Exurban**		**All villages**
	Total	Unique	Total	Unique	Total	Unique	
Other	11	3	18	5	12	4	23
Edible	54	5	79	20	70	11	100
Medicinal	15	3	20	3	10	3	27
Ornamental	92	30	109	41	39	7	152
Total	157	44	203	69	117	22	278

Table 3.3: Alpha and beta diversity estimations. The jack-knife estimate is analytically derived based on observed data, while the rarefaction estimate is based on extrapolated curves for 90 gardens in each region. β diversity based on power law relationship of extrapolated rarefaction curves ($y=Cx^z$), where z is a proxy for β diversity. Values can range from 0-1, with higher values indicating fewer overlapping species between gardens.

Urbanization region	Jack-knife estimated asymptotic diversity	Rarefaction extrapolation estimate	Exponent z (β diversity)
All species (**Fig 4; 5a**)			
All gardens (n=104)	371 (+/- 11)	337 (+/- 26)	0.4575
Suburban (n=30)	236 (+/- 16)	240 (+/- 38)	0.5934
Peri-urban (n=45)	269 (+/- 9)	244 (+/- 21)	0.4568
Exurban (n=29)	165 (+/- 11)	159 (+/- 27)	0.475
Edible species (**Fig 4; 5b**)			
All gardens	120 (+/-5)	112 (+/-12)	0.3277
Suburban	72 (+/- 4)	69 (+/-16)	0.4569
Peri-urban	99 (+/- 4)	91 (+/- 9)	0.3434
Exurban	91 (+/-6)	83 (+/-14)	0.3708
Ornamental species (**Fig 4; 5c**)			
All gardens	207 (+/- 9)	188 (+/- 21)	0.5341
Suburban	146 (+/- 12)	153 (+/- 33)	0.6721
Peri-urban	148 (+/-7)	134 (+/-17)	0.5414
Exurban	61 (+/- 7)	65 (+/- 22)	0.6436

Figures

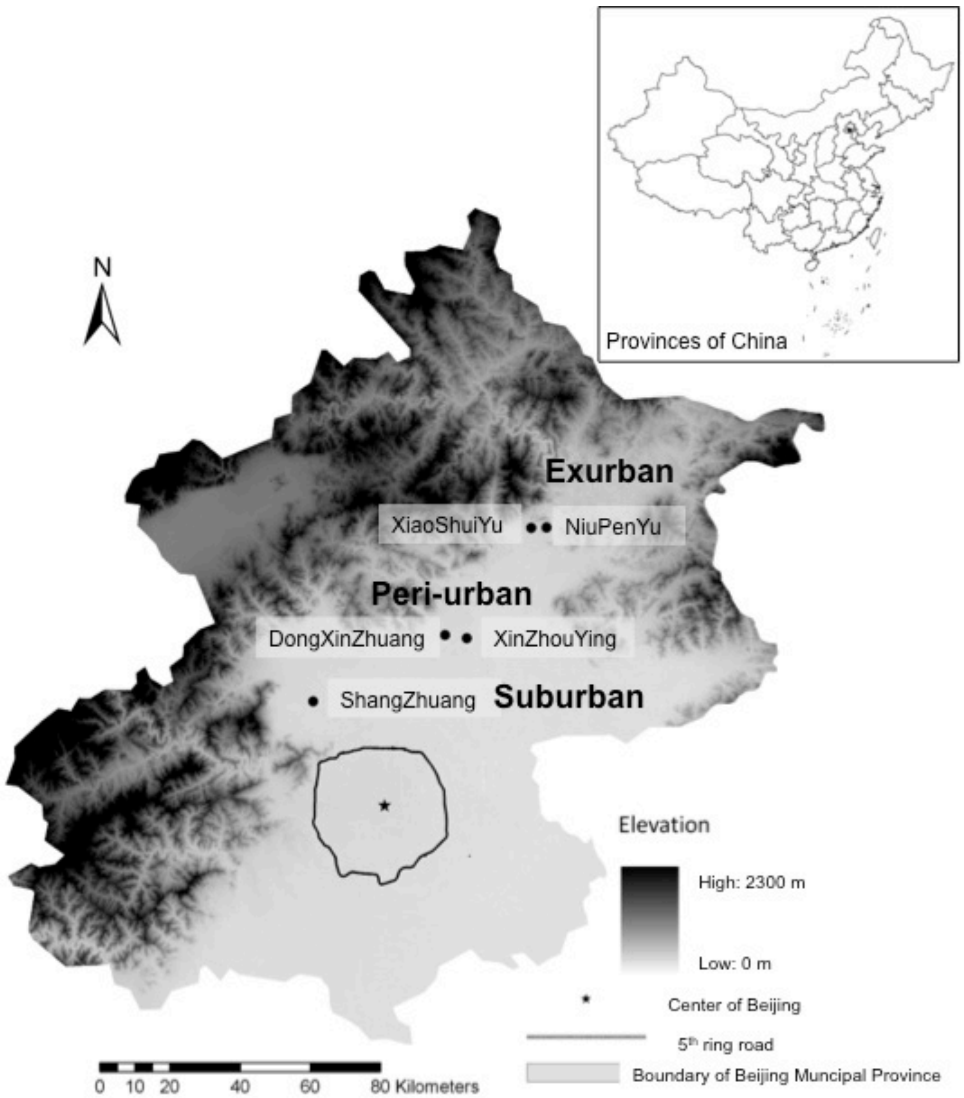

Figure 3.1: Map of Beijing municipal district indicating locations of sampled villages in Beijing Municipal Province. Inlet map shows borders of all provinces in China, with Beijing noted as a star. Elevation is indicated through shading. The star indicates the center of Beijing and the circle around it is the urban border (the 5th ring road). Circles indicate villages, and urbanization regions are indicated in bold text near villages.

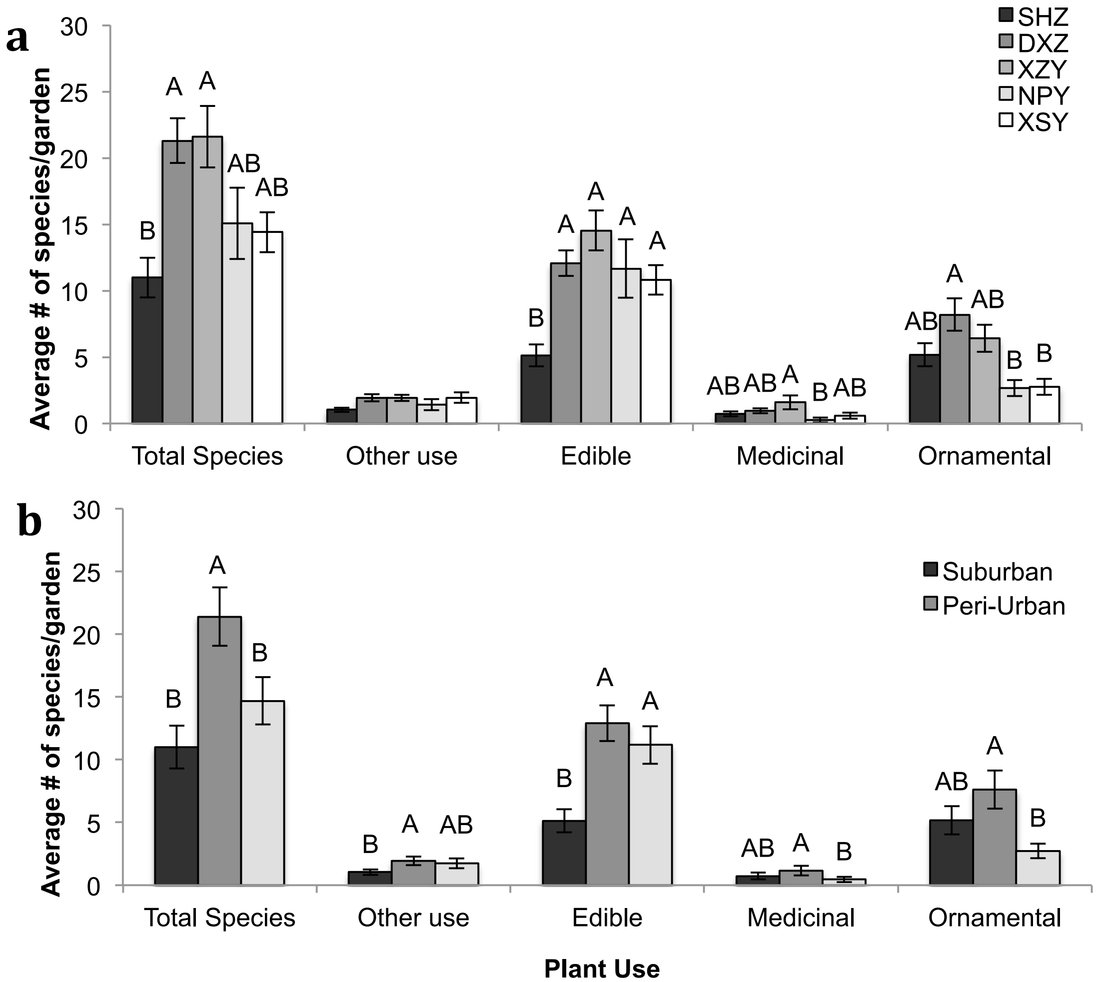

Figure 3.2: Average number of species per garden in villages (a) and urbanization regions (b), separated into plant uses. Different letters in each use denote significant differences between average number of species/garden in each village or region ($p<0.05$). Error bars represent standard error.

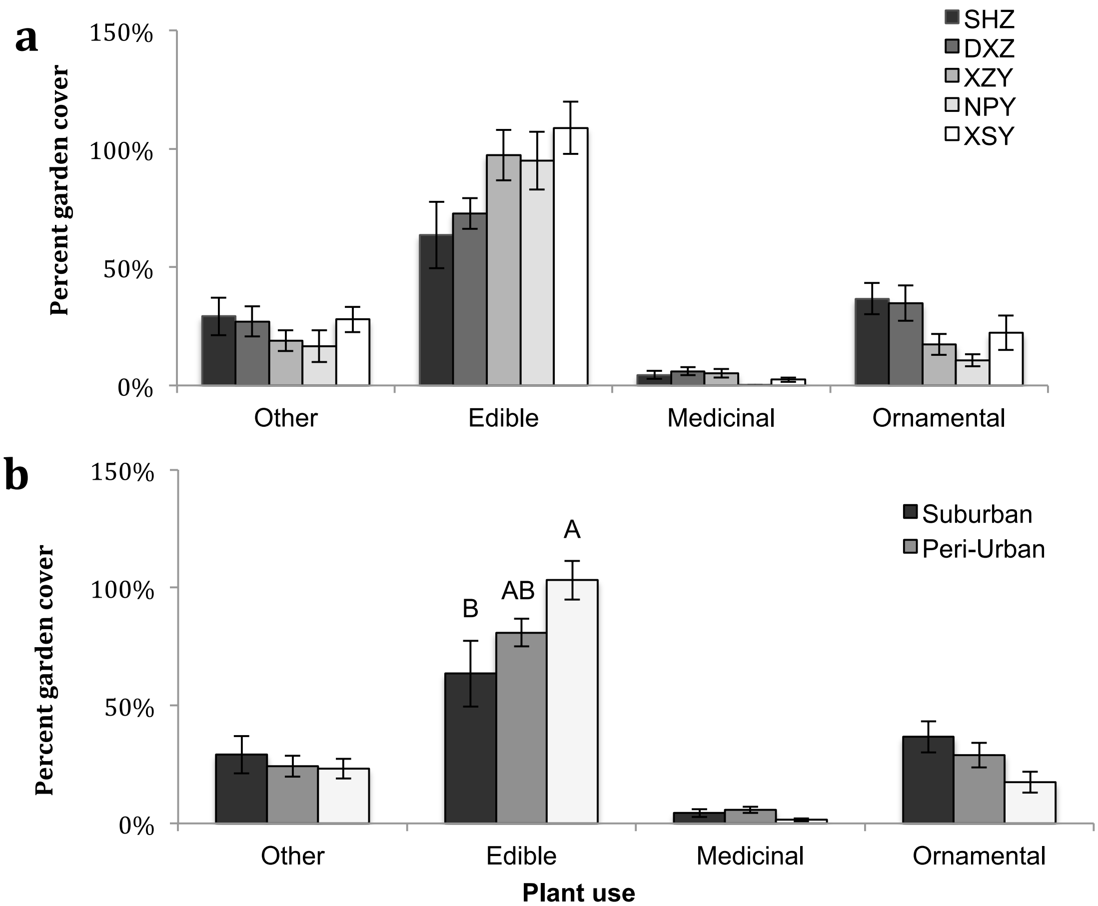

Figure 3.3: Average percent species cover in villages (a) and urbanization regions (b), separated into plant uses. Different letters in each use denote significant differences in garden cover for that use at that scale (p<0.05). At the village scale (a), there were no significant cover differences within use categories. Error bars represent standard error.

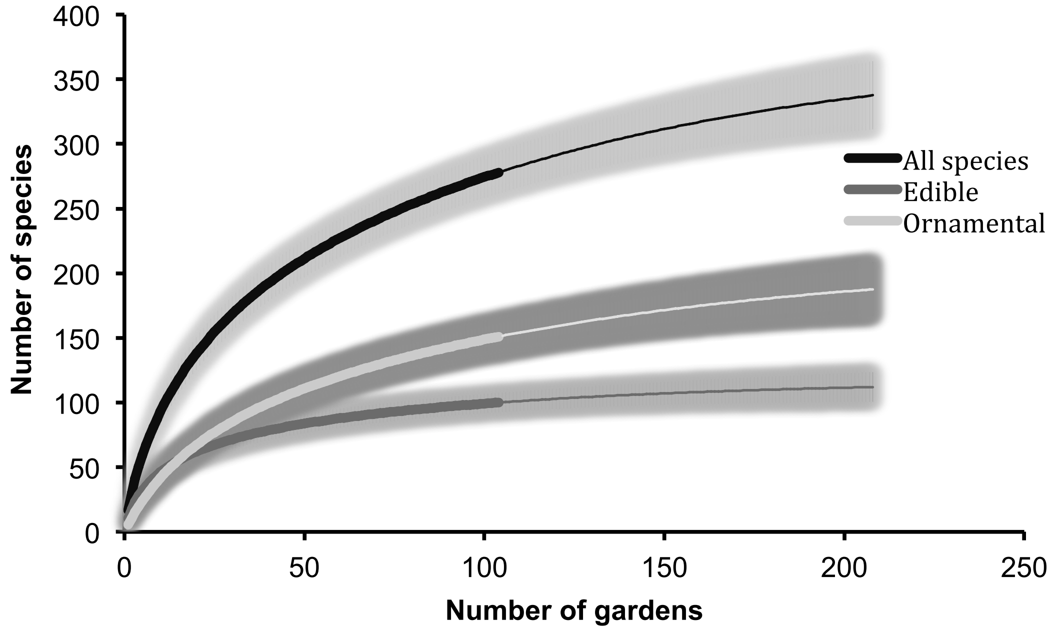

Figure 3.4: Sample-based rarefaction curves for all 104 visited gardens. One curve represents all species found (black line), and others represent the most common species uses: Edible (dark grey) and Ornamental (light gray) species. Each curve has been extrapolated to 208 gardens using EstimateS 9.0 (Colwell 2012); thick lines indicate observed gardens and thin lines of the same color represent extrapolation.

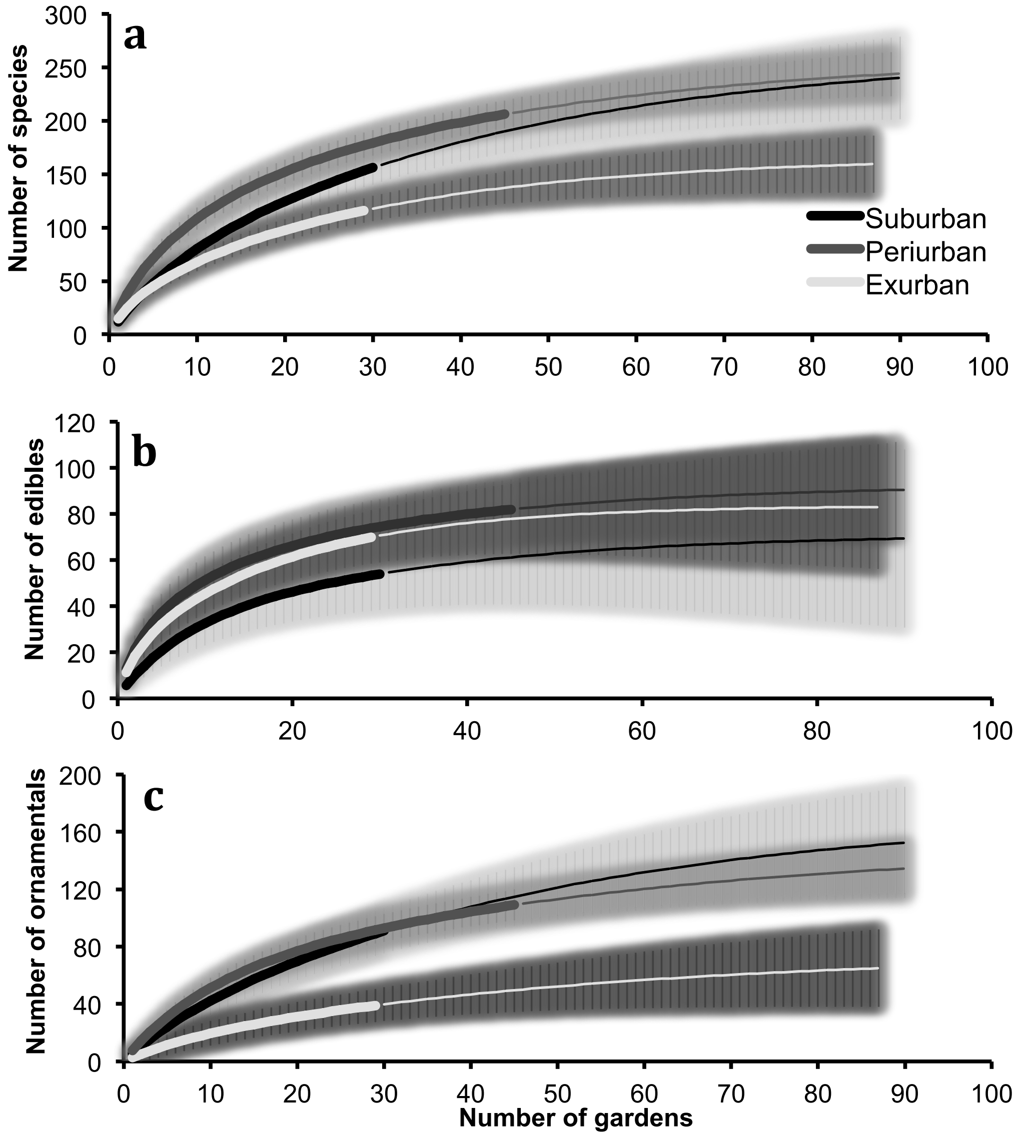

Figure 3.5: Each panel represents one of the rarefaction curves from Figure 3.4 divided into the three urbanized regions: All species (a), edible species (b), and ornamental species (c). Within each panel, suburban gardens= black line; peri-urban= dark grey line; exurban= light grey line. Shaded regions surrounding each line represent 95% confidence levels. Each curve has been extrapolated to 90 gardens. Solid lines indicate observed patterns and dotted lines indicate extrapolation. When confidence levels do not touch, that region is significantly different in species diversity than other regions.

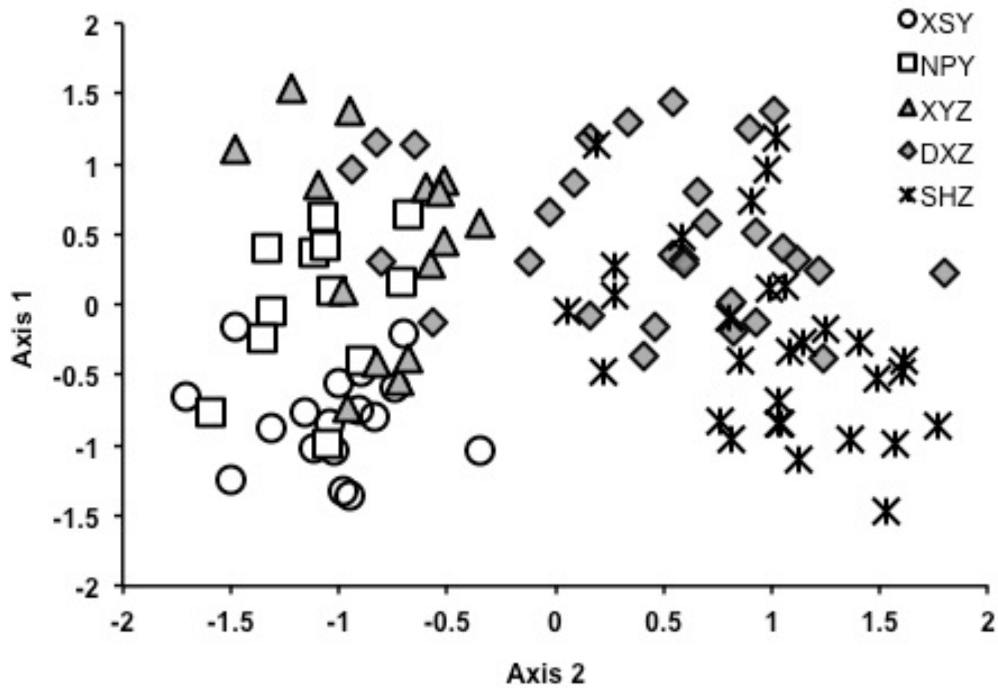

Figure 3.6: Principal components ordination of garden scale biodiversity between villages. Each point represents the biodiversity of one garden. Exurban gardens are white circles and squares; Peri-urban gardens are gray triangles and diamonds; suburban gardens are black stars. The two axes plotted account for only 12% of the variation between gardens, due to the large variation in species.

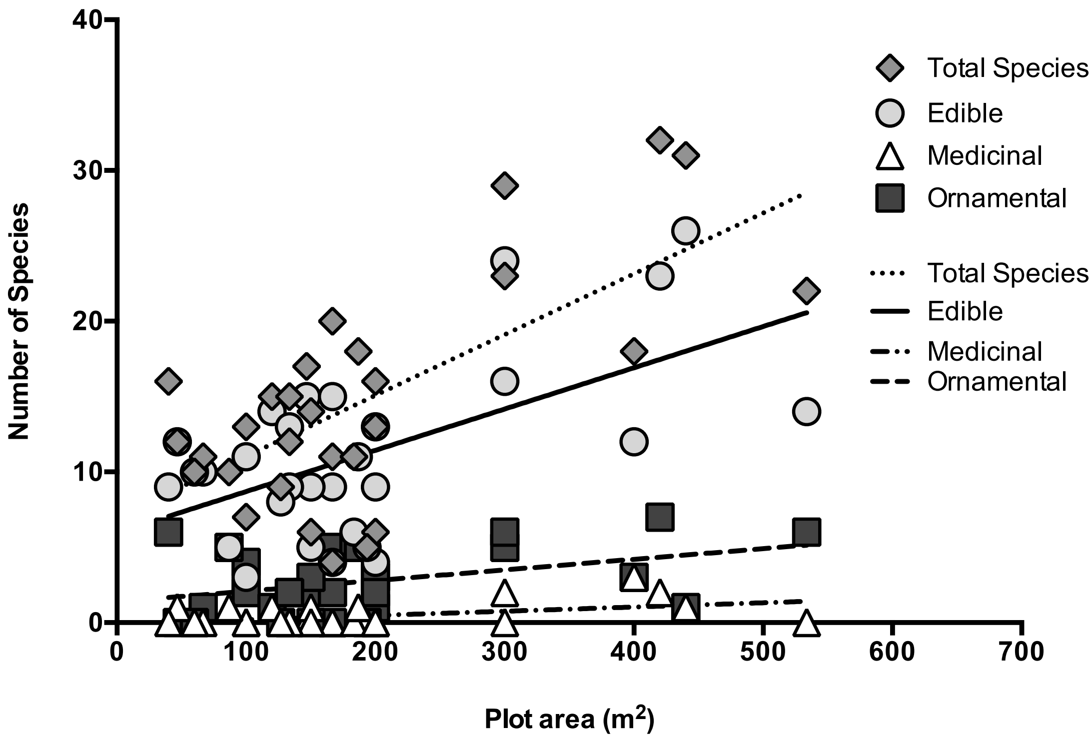

Figure 3.7: Number of species per garden as a function of garden area. Only exurban gardens are depicted here. Garden area is positively related to total number of species (diamonds; r^2=0.440, p<0.001), number of edibles (circles; r^2=0.337, p<0.001), number of ornamentals (squares; r^2=0.154, p=0.035), and number of medicinals (triangles; r^2=0.196, p=0.016) in each garden. No other villages or regions showed species-area relationships.

Chapter 4: Effects of the urban environment and soil management on the concentration and speciation of trace metals in Los Angeles urban agricultural soils

Abstract:

Heavy metals in urban soils poses a human health risk, especially in urban gardens where metals may be taken up into crops. Understanding metal speciation and bioavailability can inform managers about exposure risks and contamination sources. I ask two questions: *What factors influence the presence of heavy metals across community gardens in Los Angeles and how do these vary between Pb, As, and Cd?* and *What management and soil characteristics influence bioavailability of metals and how do these vary between Pb, As, and Cd?* I sampled cultivated and uncultivated soils in and around twelve community gardens in Los Angeles County, CA for the presence and bioavailability of lead (Pb), arsenic (As), and cadmium (Cd). Soils from cultivated and uncultivated spaces were tested with ICP/AES for overall concentration of Pb, As, and Cd. Sequential sampling was then done on a subset of these soils to identify whether metals were in exchangeable, reducible, organic, or residual fractions. I found that proximity to road increased concentrations of all metals, supporting air pollution deposition. I found the highest levels of Cd and As in garden soils that used to be in commercial cultivation (due to increased levels of Cd in mineral fertilizers), and had treated wood (due to leaching of As from chromated copper arsenate treatment). Reducible Pb increased with age of neighborhood, indicating contamination from oxidized lead paint. Exchangeable Cd and As increased strongly with proximity to road, indicating that fraction was linked to air pollution. As, especially exchangeable, only increased with road proximity in cultivated space, due to reaction with humic acids

releasing reducible As. Cd bioavailability was mitigated by OM, as organic compounds due to its high adsorptive capacity. These results suggest clear management techniques for reducing risk, such as removing treated wood, moving crops away from road edges, and moderating organic matter content. This paper can inform risk assessment in urban residential soils, predict metal accumulation hotspots, and aid in remediation of soil to reduce plant uptake and human exposure to accumulated metals.

Introduction

Heavy metal contamination in urban soils poses a human health risk in densely populated metropolitan regions (Nicholson et al. 2003; Yesilonis et al. 2008). The majority of metal contamination in cities is anthropogenic, deposited by air pollution and legacies of land uses and building materials (Nazzal et al. 2013; Mielke et al. 1983; Charlesworth et al. 2010; Nicholson et al. 2003). Urban gardens may act as an exposure route for gardeners, as metals may be potentially assimilated into harvested crops (Finster et al. 2004; Säumel et al. 2012; Murray et al. 2011; Khan et al. 2008). Community gardens, one type of urban agriculture, are often established in derelict portions of the landscape with little attention paid to the presence of heavy metals (Chaney et al. 1984; Sipter et al. 2008; Lawson 2005). Though these risks are evident in urban garden, most research on metals addresses presence and human exposure, not the mechanisms and management activities that affect metal bioavailability (Finster et al. 2004; Moir and Thornton 1989; Intawongse and Dean 2006: Lopes et al. 2011). In addition, understanding geochemical phases of metals and environmental factors affecting them will improve management of exposure risks (Schwarz et al. 2012; Virtanen et al. 2013; Mossop and Davidson 2003). My research project investigates mechanisms influencing presence and geochemical fractions of metals in community garden soils of Los Angeles, a major metropolitan area. I describe interactions between soil characteristics and metal bioavailability at small spatial scales, including implications for both human well-being and ecosystem processes.

I focus on three metals that contaminate soils and present a clear human health risk. lead (Pb), arsenic (As), and cadmium (Cd). The most studied metal is Pb, for its acute risk to children and women of childbearing age and its persistence in the urban environment (Schwarz et al. 2012; Mielke et al. 2010; Wang et al. 2006). Leaded paint and leaded gasoline are the major sources of soil contamination, both used extensively from 1884 (paint) or 1920 (gasoline) to the early 1980s (Kerr and Newell, 2003). Despite its neurotoxicity in children, safety levels for where children play vary greatly at the state (CA: 200 ppm; MI: 100 ppm) and federal level (400 ppm) (EPA 2005). In contrast, As is naturally high in CA soils, with background concentrations of 2-40 ppm (Diamond et al. 2009), and median levels near 5 ppm. The main anthropogenic sources of As are deposition by air pollution or leaching from treated wood (Charlesworth et al. 2008; Wilson et al. 2010; Hemond and Solo-Gabriel 2004). Urban garden vegetables in plots lined with CCA wood have been shown to contain elevated As (Sipter et al. 2008). No clear "safety" level for As in soils has been set federally, with clean-up focused on areas with high exposure risks (Baldwin and McCleary 1998). Cd is used as a stabilizer in tires and present in smaller quantities in vehicle exhaust, causing buildup of roadside pollution

(EPA 1999; Ellis and Revitt 1982; Cullen and Maldanado 2012). Median levels of Cd in California are low, ~.2 mg/kg, with a global average of 1.1 mg/kg (Alloway 1998). Cd is very bioavailable, and crop uptake is difficult to predict, as plant tissue levels may have higher overall levels than the soil itself (Murray et al. 2009).

Understanding the sources of metal contamination and mechanisms for fractionation can aid urban risks in mitigating exposure risks and identifying potentially contaminated urban spaces. Most metal contamination has been reported in proximity to roads (Yesilonis et al. 2008; Zhang 2006). Pb, As, and Cd are all expected to be deposited by automobiles, and lead paint may accumulate in roadside dusts and be kicked up by vehicular movement (Intawongse and Dean 2008; Charlesworth et al. 2010; Schwarz et al. 2012). Neighborhood age also may influence contamination. In particular, Pb may be elevated near older homes, due to the prolific use of lead paint between the years of 1886 and 1970 and persistence of Pb in soils (Schwarz et al. 2012; Yesilonis et al. 2007). Roadside traffic deposition As and Cd would be expected to increase in older neighborhoods due to prolonged exposure (Yesilonis et al. 2008; Nazzal et al. 2013). Land use and management levels may also impact metal levels. High levels of Cd in mineral phosphorous fertilizer may contribute to persistent contamination in areas with agricultural legacies (Burt et al. 2014; Cullen and Maldanado 2012; Chen et al. 2008; He and Singh 1994; Sun et al. 2014). Though phased out of treated lumber in the last 20 years, the use of chromated copper arsenate (CCA) treated wood to line garden plots, or as a legacy from previous building materials, may significantly affect As contamination (Stillwell et al. 2006; Stillwell et al. 2008).

Critical to assessing heavy metals and human exposure in the urban landscape is understanding metal mobility and geochemical cycling and their heterogeneity at small scales (Burt et al. 2014; Grzebisz et al. 2002). Heavy metals can exist in a bioavailable exchangeable fraction in the soil, in less available metal-oxide bound form (reducible), sequestered in organic compounds, and in incalcitrant complex forms (residual) (Mossop and Davidson 2002). Management activities, including tillage, addition of fertilizer or organic matter (OM), and watering can all affect the form of soil metals (Murray et al. 2011; Wilson et al. 2010). For example, additions of OM in the form of compost or manure have been shown to shift Cd from exchangeable to reducible or organic form, making it less available to plants (Sun et al. 2014; Cullen and Maldanado 2012). In contrast, higher levels of organic matter and humic acids have been shown to mobilize As out of an oxide-bound state and into exchangeable forms (Wilson et al. 2010; Murray et al. 2009). Altered pH levels may also impact availability of metals, including Pb (McClintock 2012). As pH and organic matter of soils can be directly affected by garden managers, understanding the mechanisms of contaminant mobility in soils may be one way for garden participants to reduce their exposure risks.

Factors influencing metal presence and mobility may interact in the urban environment to create complex patterns of metal availability. The constant turnover of cultivated soils may dilute legacy sources of metal. Metals that are deposited outside of automobile pollution may be in a different form. For instance, deposited Pb from lead paint is in reducible form (PbO), meaning I might expect an increase of reducible Pb in older neighborhoods. In addition, high levels of OM in cultivated areas may alter the forms of As and Cd as compared to uncultivated regions.

To address fundamental knowledge gaps about heavy metal presence and bioavailability in urban agriculture, I ask two questions: *What factors influence the presence of heavy metals across community gardens in Los Angeles and how do these vary between Pb, As, and Cd?* and *What management and soil characteristics influence bioavailability of metals and how do these vary between Pb, As, and Cd?* To answer these, I sampled cultivated and uncultivated soils in and around twelve community gardens in Los Angeles County, CA for the presence and bioavailability of Pb, As, and Cd. I evaluate the hypotheses that organic matter, distance from road, and age of neighborhood influence overall and sequential levels of Pb, As, and Cd. In addition, I investigate the difference in metal concentration and speciation between cultivated and uncultivated soils. Finally, this survey also asks whether treated wood and legacies of agriculture influence As and Cd levels, respectively. Understanding how management, land use, and pollution influences metal dynamics will be of use to individual managers, city planning departments and commercial agriculture alike.

Study area:

My study was focused on twelve community gardens across Los Angeles County, most centered near downtown Los Angeles (LA) (Figure 4.1). LA County covers 10,510 km^2 with approximately 9.8 million residents, (30% of California) and a population density of 1,000 people per km^2 (2010 U.S. Census). The residents of LA live very densely, with close to half of all housing units defined as multi-family housing. LA has a long history of habitation; it was officially founded in 1781 and by 1900 it was a major metropolis with over 100,000 people (Ríos-Bustamante 1992). The big boom of population in the early 1900s was a result of the Reclamation Act of 1902, providing federal funding for agricultural irrigation through the LA aqueduct (Monroy 1999). With the aqueduct came the greening of the previously dry LA basin and intensification of farming in the San Fernando Valley (Surls and Gerber 2010). Many regions outside of densely populated downtown were used as agriculture until their eventual development. The majority of LA's residential neighborhoods were built during two periods (1950-1965 and 1975-1990), the first period being when commercial agriculture was discontinued in LA city boundaries (Surls and Gurber 2010). Air pollution over the course of 150 years of industry has contributed heavily to metal contamination. LA today has high traffic density and seven major freeways, with over 300,000 commuters each day (annual average daily traffic) on each freeway (LA County Almanac/CalTrans 2005).

Methods

Data collection:

Soil samples were collected at each of twelve community gardens between the months of June-September in 2011 and 2012. During this time period, cultivated plots had peak crop growth. These gardens were chosen in 2010 for a biodiversity study, and come from a wide range of neighborhood age, income, and land use background (Figure 4.1). A soil borer was used to collect soil from a single region to 15 cm deep, the effective depth of crop soils (Brady and Weil, 2002), and at least five locations were chosen and homogenized for each sample. In each garden, soils were collected in three microenvironments: cultivated plots (5-10 per garden), uncultivated soil inside the garden (2-3 per garden), and uncultivated external soil (2-3 per garden). In plots, soil was sampled from beneath multiple crop canopies (if present), and from the middle and edge of the plot. Uncultivated soil samples within the garden were collected from the margins and paths between garden plots. Outside soils were collected from public, uncultivated spaces (unmaintained city right of ways and abandoned lots) and city managed green space. Care was taken to collect external soils at locations with similar exposure to roads. Commonly, the soils in public areas were too dense or compacted to sample to the full 15 cm. In these cases, I dug down to 5 cm sampled available soil. In total, I collected 74 plot soils, 40 uncultivated garden soils, and 33 uncultivated outside soils (147 samples).

Collected soil samples were dried at 70 °C for 72 hours and sieved to 10 mm. Dry samples were tested for total OM content, (i.e., loss on ignition, LOI) by combusting the dried sample in a muffle furnace at 550 °C for 4 hours. Percent OM was determined by mass difference. Soil pH values were measured using a 5:2 water to soil solution. All lab activities were completed at the University of California, Riverside. Soils were then sent to ALS Chemex for analysis of heavy metal concentrations using Inductively Coupled Plasma- Atomic Emission Spectroscopy/Mass Spectroscopy (ICP-AES/MS) for total levels of 29 metals, including Pb, As, and Cd, my target metals. Detection limits were as follows: 0.1-10,000 mg/kg (Pb); 0.2-10,000 mg/kg (As); 0.01-1000 mg/kg (Cd).

A sequential extraction for a sub-sample of 36 plot soils, 12 uncultivated garden soils, and 23 uncultivated outside soils (total n=71) was completed at the University of Pittsburgh to determine the accessibility of metal fractions to managed crops. I followed the modified BCR protocol methodology outlined by Mossop and Davidson (2003) for sequential extraction of metal fractions. The first extraction step is designed to extract cation elements, bound to mineral and organic matter surfaces by electro-static forces (exchangeable) (Virtanen et al. 2013). For the exchangeable fraction, 0.25 g sample of soil was treated with 0.11 mol acetic acid overnight. The resulting solution was spun in a centrifuge at 1500 RPM, and the resulting extract was filtered into a separate test tube. The second extraction step (reducible) is performed under reducing conditions, and aims to release trace metals sorbed onto Fe and Mn oxihydroxides. For the reducible fraction, the residue from the previous extraction was treated with 0.5 mol hydroxylammonium chloride at pH 2, and then centrifuged and filtered like the previous extraction. The final extraction step aims to break down organic complexes that immobilize trace metals in the soils. The residue from the previous treatment was treated with 30% hydrogen peroxide to break down organics and then washed with ammonium acetate to extract the organic fraction. The resulting samples for exchangeable, reducible, and organic metals was analyzed at the University of Pittsburgh using ICP-AES/MS. Totals for these sequential extractions were then compared to totals shown by ALS Chemex for a residual fraction.

Due to missing samples and contamination, some sequential data was unavailable for analysis. The reducible fraction of As was determined to be unusable, as interference from Cl in the hydroxylammonium chloride solution caused large overestimates.

Therefore, the residual fraction reported contains both incalcitrant forms of As and the fraction sorbed onto Fe and Mn. As reportedly exists mainly in the reducible fraction for most soils (Wilson et al. 2010), so I expect that the incalcitrant residual fraction will be a small portion. Two samples each from the exchangeable and reducible analysis were missing (four samples total), for a total of 69 valid samples for each extraction. Ten incomplete data points from the organic analysis meant that only 61 were useable.

Data analysis:

My goal in analyzing these data was to evaluate mechanisms of organic matter, distance from road, and age of neighborhood and how they influence overall and sequential levels of Pb, As, and Cd. I also compared patterns between cultivated spaces (plots) and uncultivated spaces (inside of gardens and unmaintained public soils). Secondarily, I also investigate whether treated wood and a history of agriculture influences As and Cd levels, respectively.

Distance from road was measured in ArcGIS 11, using GPS points of individual soil measurements measured to the center of each road. Centerlines of roads were identified using the Census Bureau's Topologically Integrated Geographic Encoding and Referencing (TIGER) database. Neighborhood age was obtained from the mean of recorded built dates from 5-10 housing structures closest to the sampled garden. Housing ages were obtained from public records accessed through a commercial real estate service (www.redfin.com). During field surveys, four gardens were noted as using treated wood to line garden plots. The previous land uses of all gardens were determined based on interviews with garden managers (for the most recent land use), and on land use surveys by Works Progress Administration between 1933-1939. These maps were compared to current satellite imagery to determine what nearby structures had changed or had not been built yet. Two of the gardens were in areas used for commercial agriculture in the 1930s, and information from gardeners indicated they stayed in agriculture until their respective neighborhoods were built in 1958.

Statistical analysis

Elemental concentrations are represented using concentration per oven-dried mass of soil (mg kg^{-1}). Because many trace metals, especially Pb, are log-normal in distribution and are very heterogeneous in their concentrations (Schwarz et al. 2012; Yesilonis et al. 2008), I used Spearman's non-linear correlation analysis to compare trace metals and their sequential extractions to each other and to potential contamination sources and soil characteristics (distance from road, age of neighborhood, organic matter content, pH), using SPSS 22. Correlations were conducted on all soils and separately for cultivated and uncultivated soils. Significant relationships were then graphically modeled using non-linear curve fitting and AIC for best model fit (GraphPad Prism 6). Models fitted and compared between linear, quadratic, exponential growth, semi-log, log-log, and power-law relationships.

To investigate whether neighborhood age and distance from road were interacting with each other, I conducted separate correlation analyses on metal relationships to distance from road in older locations (built before 1940), and age of neighborhood in soils close to the road (<30 m). I chose 1940 as a cutoff point because it was before the two major building periods in Los Angeles and was during a time that leaded gasoline and lead paint were used. Thirty meters was chosen as a cutoff point, as other studies have shown that distance from road does not affect soils strongly after that point (Schwarz et al. 2012, Ordóñez et al. 2003). I then compared these correlations and patterns to those performed on all points to see if previously observed relationships changed with focus on old neighborhoods or locations near roadsides.

Before conducting statistical analyses, I examined the data for overall outliers and outliers in each extraction and investigated mechanism. Two samples (one uncultivated garden soil and one cultivated soil) were removed from all analyses due to very high concentrations of Pb and As (upwards of 2000 mg/kg and 85 mg/kg respectively). In one garden, all sampled soil was more than 80 m from the road (other locations were between 1-50 m from the road). Thirty meters from the road is generally the cutoff for distance from road effects, I removed that garden from the distance analysis. I also found one garden with very high (>1200 mg/kg) levels of Pb outside the garden (n=3). I include these high levels of Pb in the analysis of total soil concentrations, as there are more sampled points to compare it to. In sequential analyses, only one of these high levels was analyzed. I removed that one from sequential analysis because of the reduced number of points analyzed and how different it was from other sequentially analyzed points. Finally, we identified a garden with high levels of As in two locations, very spatially close to one another. Upon interviewing the gardener who managed them, we discovered that he was using an As based pesticide, not used in any other garden. I removed those points from the As study, as they represented a unique source of contamination.

Results

Soil characteristics and overall contamination:

Soil OM was highest in cultivated plots and was related to pH. Plot soils were higher in OM than uncultivated soils outside the garden (t-test: $p<0.05$, $t=2.314$), but not different from uncultivated soils inside the garden. There was high variability in cultivated plot OM, ranging from 4-30%, with slightly lower variation in uncultivated soils (2-24%). pH levels were surprisingly steady, 6.16-8.6 in uncultivated soils (average of 7.19), and 6.6-8.14 in cultivated soils (7.38 average). No significant relationships were found between sequential or overall metal levels and pH, though OM and pH were somewhat correlated (Uncultivated: -0.397, $p=0.001$; cultivated: -0.305, $p=0.007$).

After initial analysis of soils, I noticed that distance from road and age analyses were significant for As and Cd in some gardens or groups of gardens, but not others. For As, plot soils and internal uncultivated soils in gardens using treated wood had high levels of As inside the gardens, interfering with the distance from road gradient (Figure 4.3B). Two of these same treated wood gardens were built in 1958, and showed a high concentration of As in relatively young gardens, removing age significance (Figure 4.3A). Similarly, the largest Cd concentrations (2.5-4.3), were found in the two gardens previously used as commercial agricultural land until 1958 (Figure 4.3C). Analyses after removal of these points showed expected log-normal relationships with age of neighborhood and distance from road for Cd and As, indicating that treated wood and agriculture have some noticeable effect on garden soil contamination.

Pb levels were highly variable, (18-1720 mg/kg) and 23 soils in and around 9 gardens exceeded the recommended 200 mg/kg child exposure level, 3 of those in cultivated areas (Figure 4.2). Pb concentrations were twice as high in uncultivated regions (Figure 4.4). For As, values ranged between 2.5-17 mg/kg, and 56 locations in 10 gardens exceeded average CA background levels (5 mg/kg). Half of these were in uncultivated regions, and the only soils with over 10 mg/kg in cultivated soils were those with treated wood. Cd values were relatively low (.11-4.27), though they regularly exceeded average CA background levels (.2-1.1 mg/kg, Alloway 1998). Seventy-five soils across ten gardens had more than 1 mg/kg Cd. This was particularly noticeable in the two gardens which were previously used as commercial agriculture through the 1950s. These legacy agriculture gardens had five soils with above 2.5 mg/kg Cd.

Similar sources for these contaminants was indicated through Pb, As, and Cd overall levels all being significantly correlated with each other (Table 4.1). The strongest relationships were between Pb and Cd (0.541, $p<0.001$) and between Cd and As (0.464, $p<0.001$). The lowest correlation was still significant, between Pb and As (0.338, $p<0.001$).

Sequentially, Pb was the least bioavailable, As was more mobile, and Cd was the most bioavailable. Pb was most commonly found in the reducible fraction (42%), then residual (35%), then exchangeable and organic (Table 4.2). As was primarily in the residual fraction (69%), which contains both the reducible fraction and incalcitrant residual fraction. The next highest fraction of As was exchangeable and it was lowest in organics. Cd was very bioavailable, with between 50-90% in exchangeable in each garden, with a lower level in reducible and organic. Residual Cd was almost non-existent, due to low levels of Cd and slight variations in detection levels. For many gardens with less than 2 mg/kg overall Cd, sequential levels often exceeded the total, indicating that residual was ~0.

Age

Overall Pb decreased as year built increased (Figure 4.4-A.1), in both cultivated (-0.452, r^2=0.119, $p<0.001$) and uncultivated soils (-0.458, r^2=0.124, $p<0.001$). For sequential fractions (Figure 4.4-A.2), both reducible (-0.434, r^2=0.171, $p<0.05$) and exchangeable Pb (-0.350, r^2=0.096, $p<0.05$) decreased with younger buildings in uncultivated soils. In cultivated areas (Figure 4.4-A.3), only reducible lead decreased (-0.366, r^2=0.116, $p<0.05$). The residual fraction showed no clear change across years.

When gardens with treated wood were excluded, overall As decreased from older to younger neighborhoods in cultivated areas (-0.659, r^2=0.392, $p<0.001$), not uncultivated (Figure 4.4-B.1). The decrease in cultivated As levels appeared to come exclusively from decreased exchangeable concentrations (Figure 4.4-B.2; -0.504, r^2=0.240, $p<0.05$). There was no significant fractional change in uncultivated As levels.

Overall Cd decreased with increased built date of neighborhoods (Figure 4.4-C.1), both in cultivated (-0.370, r^2=0.248, $p<0.01$) and uncultivated (-0.536, r^2=0.298, $p<0.001$)

soils, though uncultivated is more significant. The only sequential fraction that changed was an exchangeable increase with age in uncultivated soils (-0.404, r^2=0.175, $p<0.01$).

Distance

Overall Pb decreases with distance from road in both uncultivated (-0.421, r^2=0.046, $p<0.01$) and cultivated (-0.344, r^2=0.055, $p<0.01$) soils, though the pattern is clearer in uncultivated. Only uncultivated soils (Figure 4.5-A.3) showed significant sequential changes, as exchangeable (-0.593, r^2=0.327, $p<0.01$), reducible (-0.560, r^2=0.308, $p<0.01$), and residual (-0.406, r^2=0.137, $p<0.05$) fractions decreased with distance from road.

Total As does not change with distance from road (Figure 4.5-B.1). In cultivated soils (Figure 4.5-B.2), exchangeable (-0.650, r^2=0.526, $p<0.01$) and organic (-0.594, r^2=0.580, $p<0.01$) levels increase near the road. No sequential patterns are seen in uncultivated soils.

Overall Cd only decreases with distance from road in cultivated areas (Figure 4.5-C.1; -0.370, r^2=0.208, $p<0.01$). Cd reducible (-0.641, r^2=0.347, $p<0.01$) and organic (-0.609, r^2=0.373, $p<0.01$) fractions increase in proximity to roads in cultivated soils (Figure 4.5-C.2). In uncultivated soils (Figure 4.5-C.3), only Cd exchangeable significantly decreases with distance from road (-0.501, r^2=0.113, $p<0.01$).

I also investigated whether there was an intensification of gradient effects in Pb, As, or Cd if age analyses only included locations close to the road and if distance analyses only included older neighborhoods. Contrary to my predictions, I found no distance from road effects at all for As and Cd concentrations in uncultivated areas in old regions, and nearly the same results for Pb. In cultivated areas, I found some intensification of As concentrations for distance from road analyses in older areas (organic fraction related -0.706 instead of -0.594), but the remaining results were very similar. Similarly, correlations in the age analysis which only contained soils near the road were similar to the original results (deviation of 10%). These results indicate there is no interactive effect of age of neighborhood on distance from road or vice versa.

Organic matter

The higher levels of As and Cd in locations with treated wood and agricultural background did not influence the observed relationships of OM, so I did not exclude those locations from OM analyses. While OM showed significant relationships with overall Pb (Uncultivated: -0.421, $p<0.01$) and organic Pb (Uncultivated: -.575, $p<0.01$; Cultivated: -.667, $p<0.001$), all relationships were negative. Because Pb is not being actively deposited in the soil, this pattern likely indicates that added OM just dilutes the present Pb, not actively reacts with Pb.

Overall As did not change with OM, though proportions of extractions did, indicating an interactive effect of OM with bioavailability (Figure 4.6A). In cultivated soils, exchangeable (0.665, $r^2=0.484$, $p<0.001$) and organic As (0.548, $r^2=0.042$, $p<0.001$) increase with increasing organic matter (Figure 4.6-A.2). Uncultivated soils (Figure 4.6-A.3) only have a significant increase in organic As (0.595, $r^2=0.297$, $p<0.01$).

Cd increases with organic matter in uncultivated regions (Figure 4.6-B.1; 0.652, r^2=0.241, p<0.001). This increase is from exchangeable (0.652, r^2=0.222, p<0.001) and reducible Cd (0.652, r^2=0.384, p<0.001) increasing with added OM (Figure 4.6-B.3). Only reducible Cd (0.520, r^2=0.225, p<0.001) increases in cultivated areas with added OM (Figure 4.6-B.2).

Discussion

This study demonstrates clear mechanisms for proliferation and speciation of metals in metropolitan regions. I show widespread and highly variable anthropogenic metal contamination across sampled gardens (Figure 4.2). Overall, cultivated locations are less contaminated than uncultivated, likely due to tillage and addition of new soil diluting metal concentrations. Other studies support these main results showing metal concentration relationships to distance to road and year built (Wang et al. 2006; Yesilonis et al. 2008), though mine is the first to show how metal speciation changes with mechanisms. In particular, reducible lead proliferation in older neighborhoods (Figure 4.4A) indicates contamination by lead paint (PbO). Distance from road is the most explanatory for Pb, As, and Cd (Figure 4.5), with this common source supported by their correlation with each other (Table 4.1). In addition, complex speciation of Cd and As (Figure 4.6) influenced by interaction with OM may explain why some distance from road gradients are stronger in cultivated spaces. Gardeners have the most control over OM and pH changes in their soil and may be able to mitigate available Cd and As with management techniques. My study on urban garden metal dynamics provides valuable results to understand interactions between gardeners, environmental processes, anthropogenic pollution and the resulting consequences for soil contamination.

Overall metal presence

Many samples of soils had metal concentrations above background and safety levels (Figure 4.2), which is consistent with anthropogenic pollution. Pb is the least available to crops (most available in reducible form), but shows the most extreme variability and contamination above established safety levels of 200 ppm (Figure 4.2, Table 4.2). No safe levels of Pb have been established in agricultural soils, especially for sensitive populations, like pregnant women and young children, as soil ingestion and breathing contaminated dust can increase blood Pb levels to unsafe levels (Schwarz et al. 2012; Mielke 1997; Lanphear et al. 2000, Koller et al. 2004). Hyper-accumulator plants, like brassicaceous species, send out exudates which can release reducible Pb into the exchangeable fraction, increasing exposure through plant species (Murray et al. 2011). Mustards and other brassicaceous species are commonly planted in community gardens. As also exists primarily in the reducible segment, which can also be taken up by hyper-accumulators. Cd is the most bioavailable of investigated metals, and is mostly in exchangeable form (Table 4.2). Cd uptake by leafy or root vegetables is difficult to predict, as even small concentrations in the soil can be taken into plant tissue at a high rate (Murray et al. 2009; Murray et al. 2011; Alloway et al. 1998). Because of its exchangeable affinity, long-term ingestion of plants grown in Cd contaminated soils may be of higher concern than Pb (Charlesworth et al. 2010).

These results show that As and Cd are deposited in more ways than just air pollution. Gardens using treated wood have cultivated soils with As levels (~6.5 mg/kg) that rival levels of soils directly bordering major roads (~5.5 mg/kg) (Figure 4.3B). These concentrations are consistent with chromium copper arsenate (CCA) leachates used to preserve wood (Alloway 1998; Brandstetter et al. 2000), which create a local (less than .5 m radius) but intense contamination. Though my study did not contain enough replication to comparatively analyze treated and untreated gardens, CCA wood is a major source of As in urban areas (De Miguel 2007). Cd levels were elevated in and around two gardens that were in commercial agricultural cultivation until 1958 (Figure 4.3C). Mineral fertilizers used extensively in commercial agricultural land can contain high levels of Cd, due to its affinity to adsorb to rock-bound phosphates (Andresen and Küpper 2013; He and Singh 1997). Cd levels in mineral fertilizer have been reduced greatly in the past 20 years (Cook and Morrow 1995), but long-term legacies of deposited Cd can persist in the soil (Orroño and Lavado 2009).

Age and distance from road gradients

Understanding the sources of anthropogenic pollution is important to assessing risk factors for gardener exposure. Older structures near gardens and proximity to roads both increased Pb concentrations, though sequential analyses indicate that contamination sources are different for each mechanism. Gardens near older buildings have markedly higher levels of Pb, even in cultivated regions (Figure 4.4-A.1). Though other studies support this result (Schwarz et al. 2012; Yesilonis et al. 2008; Wu et al. 2010), mine is the first to quantify how neighborhood age affects Pb fractionally (Figure 4.4-A.2, A.3). Background Pb levels, from mineral concentrations, reside in the residual fraction, very unavailable to plants (Säumel et al. 2010). These results indicate that the increased Pb in older neighborhoods is almost exclusively in the reducible fraction (Figure 4.4B, C). This legacy signal likely comes from lead paint, which comes in lead tetraoxide (Pb_3O_4) form, a reducible compound (Boreiko and Battersby, 2008). Lead paint was used on homes and buildings in the U.S. from 1860 to 1976, and older homes tend to have been repainted or have flaky lead paint, contributing to soil pollution (Jacobs et al. 2002). In contrast, I show that exchangeable, reducible, and residual Pb all decrease with distance from road (Figure 5A.3) with a sharp decrease after 20 meters. Exchangeable, or bioavailable, Pb is usually associated with major roads (Wu et al. 2010) and residual Pb may persist in soils and road dust for centuries (Mielke 1997; Elless et al. 2007). Pb pollution in association with roads is likely from legacies of leaded gasoline deposition, used in automobiles from 1912-1980 (Kerr and Newell 2003). Cultivated soils are changed out consistently through tillage and addition of amending compounds like compost and do not show an effect of proximity to roads, though the age effect on reducible Pb is still evident (Figure 4.5-A.2, Figure 4.4-A.2). This indicates that air deposition is no longer occurring, though Pb is still actively being added to soils through degradation of leaded paint in older neighborhoods.

As is also related to neighborhood age and distance from road, though sequential analyses indicate that these relationships come from the same pollution source. Increase in As in older neighborhoods and with proximity from road is almost exclusively exchangeable (Figure 4.4-B.2, Figure 4.5-B.2). Air pollution deposition from car exhaust is generally in the exchangeable fraction (Haygarth and Jones 1992), which would explain both observed age and distance from road gradients. Older regions have been exposed to roadside pollution for longer time periods, and areas more distant from the road have been exposed less. Surprisingly, these significant relationships are only seen in cultivated areas, even though the highest concentrations occur in uncultivated regions. Cultivated soils may better show age and road effects because As fractions are more reactive in high OM soils (Murray et al. 2011; Wilson et al. 2010; Figure 4.6-A.2). Increased tillage and water addition may further facilitate As transformations. As may be added in to the edge of the garden and react with humic acids to increase the exchangeable fraction and then be sequestered by organic matter in plots (Figure 4.5-B.2). The uncultivated soils I sampled mostly ended up very near the road. Though these locations were high in As, the lack of range may have prevented clear correlations.

Similarly to As, Cd shows increased exchangeable concentrations in older areas and with proximity to roads (Figure 4.4C, 4.5C). Established road contamination of Cd comes from tire residue and older neighborhoods have long-term exposure (Lagerwerff and Specht 1970). Uncultivated areas show the most significant increases in Cd with road proximity (Figure 4.4-C.3). This indicates that dilution from OM and soil tillage may reduce Cd contamination. Increased OM in cultivated areas may better sequester deposited exchangeable Cd by adsorbing it and trapping it in organics (Sun et al. 2014; Sauve et al. 2003). This process is likely, due to the increase in reducible and organic Cd in cultivated soils near the road.

Organic matter in cultivation

Higher levels of OM in cultivated soils doesn't change overall levels of Cd, but may move exchangeable Cd into the reducible fraction through adsorption onto organic molecules, making it less available to plant roots (Figure 4.6-B.2). OM and pH are some of the most important ecosystem properties within gardener control, and cultivation practices can influence plant uptake of metals, even at small scales (Mollison 1990; Probert et al. 1995). OM has been used to remediate of metal contaminated soils (Herwijnen et al. 2004), as its adsorption capacity is up to 30 times that of clay (Sauve et al. 2003). In uncultivated areas, OM adsorbs the more mobile exchangeable Cd that is present near the roadside (hence the overall increase of Cd with added OM outside cultivation), but exchangeable levels remain high. These results support recent research indicating that plant uptake of Cd can be mitigated through addition of fertilizer (Sun et al. 2014).

In contrast, my results indicate an increase in bioavailability of As in high OM cultivated regions. Organic As also increases, likely because of the easily sequestered As compounds released into soils (Figure 4.6-B.2). Although As sorbtion onto organic matter has been recorded in other studies, it did so at lower pH than found in this study (Thanabalasingam and Pickering 1986). This release of bioavailable As was likely caused by reaction of humic acids with Fe and Mn oxides, releasing adsorbed reducible As compounds into exchangeable fractions (Wilson et al. 2010; Wenzel 2013). Compost and manure, added to cultivated soils, have more humic acids than OM found in uncultivated areas (Murray et al. 2011), encouraging the mobility of As in managed soils.

Management recommendations

The results from this comprehensive metal presence and sequential extraction study can inform both the structure and location of future urban gardens, as well as reduce risk factors for current gardeners. My findings suggest clear management activities, which will reduce risk of metal exposure and crop uptake of metals in urban gardens. Growing soils in raised beds without the use of locally contaminated soils may reduce plant uptake of soil metals, especially in older areas with elevated levels of lead paint or agricultural regions which may have elevated soil Cd (Heinegg et al., 2009). The markedly reduced Pb and Cd levels that I found in soils more than 20 meters from the road indicate that cultivated plots should be at least this far from the road for reduced amount of contamination. These structural suggestions are also supported by EPA literature (Turner 2009). OM was shown to reduce mobility of Cd, the highest risk for plant uptake, though elevated OM may mobilize As. A risk management suggestion for garden participants would be to moderate OM percentages. Crops only need 5-15% OM for optimum growth (Ontario Ministry of Agriculture and Food, 2009), and gardens with higher levels may risk unnecessary mobility of As compounds. In addition, the recorded increase in As concentration where treated wood was used should encourage gardeners who have CCA treated wood lining their plots to remove it and replace it with uncontaminated materials.

Conclusion:

This research highlights the various anthropogenic sources of Pb, As, and Cd and their interaction with management activities. In particular, I found the strongest relationships between distance from road and distribution of metals. For Pb, neighborhood age increased reducible concentrations, indicating a legacy of lead paint, while exchangeable and other extractions increased with proximity to roads. This indicates that air pollution and persistent lead dust were the source for distance from road based relationships. As and Cd contamination was mainly in exchangeable form, but showed complex relationships with cultivation activity. Exchangeable forms of both metals, likely deposited from air pollution, accumulated near the road and with age of neighborhood. Cd became more diluted in cultivated area due to tillage practices, while As was intensified, due to the contrasting ways these metals react to addition of OM. Finally, I show how built structures, like treated wood, and commercial agricultural history can affect As and Cd levels, influencing possible plant uptake. Through the sequential analysis in cultivated and uncultivated regions, scientists can begin to understand how landscape, legacies, pollution, and management activities interact to create metal dynamics in urban agricultural regions. This knowledge is essential for risk assessment in urban residential soils, predicting metal accumulation hotspots, and remediating soil to reduce plant uptake and human exposure to accumulated metals.

Works Cited:

Alloway, B.J. (ed.) 2013. Heavy Metals in Soils: Trace Metals and Metalloids in Soils and their Bioavailability (third edition). Environmental Pollution 22

Andresen, E. and Küpper, H. 2013. Cadmium toxicity in plants. In *Cadmium: From Toxicity to Essentiality.* (A. Sigel, H. Sigel, and R.K.O. Sigel eds.): Springer Finance 395-413 (Metal Ions in Life Sciences)

Baldwin L. and McCleary H. 1998. Study of State Soil Arsenic Regulations. Online available at <http://www.rtenv.com/PADEP/attachment3.pdf>

Boreiko, C. and Battersby, R. 2008. Voluntary risk assessment report on lead and some inorganic lead compounds. Lead Development Association International (LDAI).

Brady, N.C., and Weil, R.R. 2002. *The Nature and Properties of Soils,* 13th ed. Prentice Hall, Upper Saddle River, New Jersey, USA.

Brandstetter, A., Lombi, E., and Wenzel, W. 2000. Arsenic-contaminated soils: I. Risk assessment. In *Remediation of hazardous waste contaminated soils*, (D. Wise, H. Trantolo, H. Inyang, & E. Cichon Eds.) pp. 715–737. New York: Marcel Dekker Inc.

Burt, R., Hernandez, L., Shaw, R., Tunstead, R., Ferguson, R., and Peaslee, S. 2014. Trace element concentration and speciation in selected urban soils in New York City Environmental Monitoring Assessmenet 186:195–215

Chaney, R.L., Sterret, S.B., Mielke, H.W., 1984. The potential for heavy metal exposure from urban gardens and soils. In: Preer, J.R. (Ed.), Proceedings of the Symposium on Heavy Metal in Urban Gardens. University of the District of Columbia Extension Service, Washington, DC, USA, pp. 37–84.

Charlesworth, S. M. 2010. A review of the adaptation and mitigation of global climate change using sustainable drainage in cities. Journal of Water and Climate Change **1**:165-180.

Chen, W., Krage, N., Wu, L., Pan, G., Khosrivafard, M., Chang, A.C. 2008. Arsenic, Cadmium, and Lead in California Cropland Soils: Role of Phosphate and Micronutrient Fertilizers. Journal of Environmental Quality 37:689–695

Cook, M. E., and Morrow, H. 1995. "Anthropogenic Sources of Cadmium in Canada," National Workshop on Cadmium Transport Into Plants, Canadian Network of Toxicology Centres, Ottawa, Ontario, Canada.

Cullen, J.T. and Maldanado, M.T. 2013. Biogeochemistry of Cadmium and Its Release to the Environment. In *Cadmium: From Toxicity to Essentiality.* (A. Sigel, H. Sigel, and R.K.O. Sigel eds.): Springer Finance. pp 31-58 (Metal Ions in Life Sciences)

Diamond, D., Baskin, D., Brown, D., Lund, L., Najita, J., Javandel, I. 2009. Analysis of Background Distributions of Metals in the Soil at Lawrence Berkeley National Laboratory. Lawrence Berkeley National Laboratory Environmental Restoration Program

U.S. Environmental Protection Agency. 1999 Integrated Risk Information System (IRIS) on Cadmium. National Center for Environmental Assessment, Office of Research and Development, Washington, DC.

Elless, M.P., Bray, C.A., Blaylock, M.J., 2007. Chemical behavior of residential lead inurban yards in the United States. Environmental Pollution 148, 291-300.

Ellis, J. B., and Revitt, D. M. 1982. Incidence of heavy metals in street surface sediments: solubility and grain size studies.Water, Air, and Soil Pollution 17:87–100

Finster, M.E., Gray, K.A., and Binns, H.J.. 2004. Lead levels of edibles grown in contaminated residential soils: a field survey. Science and Total Environment 320:245–257

Grzebisz, W., Ciesla, L., Komisarek, J., and Potarzycki, J. 2002. Geochemical assessment of heavy metals pollution of urban soils. Polish Journal of Environmental Studies, 11(5), 493–499.

Haygarth, P.M., and Jones, K.C. 1992. Atmospheric deposition of metals to agricultural surfaces. In: Biogeochemistry of trace elements (Adriano DC eds). Boca Raton: Lewis Publishers, p. 249–276.

He, Q.B. and Singh, B.R. 1994. Crop uptake of cadmium from phosphorus fertilizers: II. Relationship with extractable soil cadmium. Water Air and Soil Pollution 74:251–265.

Heinegg, A., Maragos, P., Mason, E., Rabinowicz, J., Straccini, G., and Walsh, H. 2000. *Soil contamination and urban agriculture: A practical guide to soil contamination issues for individuals and groups*. Quebec, Canada: McGill University, McGill School of Environment. Online available at: http://www.ruaf.org/sites/default/files/guide%20on%20soil%20contamination.pdf .

Hemond, H. F. and Solo-Gabriele, H. M. 2004. Children's exposure to arsenic from

CCA-treated wooden decks and playground structures. Risk Analysis, 24, 51–64.

Herwijnen R., Hutchings, T.R., Al-Tabbaa, A., Moffat, A.J., Johns, M.L., and Ouki, S.K. 2007. Remediation of metal contaminated soil with mineral-amended composts. Environmental Pollution 150:347–354.

Intawongse, M. and Dean J.R. 2006. Uptake of heavy metals by vegetable plants grown on contaminated soil and their bioavailability in the human gastrointestinal tract. Food Additives and Contaminants 23: 36-48. Joint FAO/WHO

Jacobs, D.E., Clickner, R.P., Zhou, J.Y., Viet, S.M., Marker, D.A., Rogers, J.W., Zeldin, D.C., Broene, P., and Friedman, W., 2002. The prevalence of lead-based paint hazards in U.S. housing. Environmental Health Perspectives 110 (10) 599-606.

Kerr, S., and Newell, R.G., 2003. Policy-Induced Technology Adoption: Evidence from the U.S. Lead Phasedown. The Journal of Industrial Economics 51 (3), 317e343.

Khan, S., Caoa, Q., Zhenga, Y.M., Huanga, Y.Z., Zhu, Y.G., 2008. Health risks of heavy metals in contaminated soils and food crops irrigated with wastewater in Beijing, China. Environmental Pollution 152, 686e692.

Koller, K., Brown, T., Spurgeon, A., and Levy, L., 2004. Recent developments in low-level lead exposure and intellectual impairment in children. Environmental Health Perspectives 112 (9): 987-994.

Lagerwerff, J.V. and Specht, A.W. 1970. Contamination of roadside soil and vegetation with cadmium, nickel, lead, and zinc. Environmental Science and Technology 4(7): 583-586.

Lanphear, B.P., Dietrich, K., Auinger, P., Cox, C., 2000. Cognitive deficits associated with blood lead concentrations <10 mg/dL in US children and adolescents. Public Health Report 115: 521-529.

Lawson, L. 2005. City Bountiful; A century of community gardening in America. The Regents of the University of California. Berkeley and Los Angeles, CA, USA.

Lopes, C., Herva, M., Franco-Uria, A., and Roca, E. 2011. Inventory of heavy metal content in organic waste applied as fertilizer in agriculture: evaluating the risk of transfer into the food chain. Environmental Science and Pollution Research 18(6): 918-939.

McClintock, N. 2012. Assessing soil lead contamination at multiple scales in Oakland, California: Implications for urban agriculture and environmental justice. Applied Geography 35:460-473

Mielke, H.W., Anderson, J.C., Berry, K.J., Mielke, P.W., Chaney, R.L., Leech, M., 1983. Lead concentrations in inner-city soils as a factor in the child lead problem. American Journal of Public Health 73 (12): 1366-1369.

Mielke, H. W., Dugas, D., Mielke, P.W., Jr, Smith, K. S., and Gonzales, C. R. 1997. Associations between soil Pb and childhood blood Pb in urban New Orleans and rural La- fourche Parish, Louisiana. Environmental Health Perspectives 105(9): 950–954.

Mielke, H.W., Laidlaw M.A.S, and Gonzales, C. 2010. Lead (Pb) legacy from vehicle traffic in eight California urbanized areas: Continuing influence of lead dust on children's health. Science of the Total Environment 408 (19): 3965-3975.

Moir, A.M., and Thornton, I. 1989. Lead and cadmium in urban allotment and garden soils and vegetables in the United-Kingdom. Environmental Geochemistry and Health 11(3-4): 113-119.

Mollison, B. 1990. Permaculture: A practical guide for a sustainable future. Island Press, WA. 579 pp.

Monroy, D. 1999. *Rebirth: Mexican Los Angeles from the great Migration to the Great Depression.* University of California Press Ltd. London, England.

Mossop, K. F. and Davidson, C. M. Comparison of original and modified BCR sequential extraction procedures for the fractionation of copper, iron, lead, manganese and zinc in soils and sediments. *Analytica Chimica Acta* 2003, *478*, 111–118.

Murray, H., Thompson, K., and Macfie, S.M. 2009. Site and species-specific patterns of metal bioavailability in edible plants. Botany 87:702–711

Murray, H., Pinchin, T.A., and Macfie, S.M. 2011. Compost application affects metal uptake in plants grown in urban garden soils and potential human health risk. Journal of Soils and Sediments 11(5): 815-829.

Nazzal, Y., Rosen, M.A., and Al-Rawabdeh, A.M. 2013. Assessment of metal pollution in urban road dusts from selected highways of the Greater Toronto Area in Canada. Environmental Monitoring Assessment 185:1847–1858

Nicholson, F. A., Smith, S. R., Alloway, B. J., Carlton-Smith, C. and Chambers, B. J. 2003. An inventory of heavy metals inputs to agricultural soils in England and Wales. Science of the Total Environment 311:205-219.

Ontario Ministry of Agriculture and Food. 2009. Soil Management: Building a Healthy Soil. Online available at: http://www.omafra.gov.on.ca/english/crops/pub811/8building.htm

Ordóñez, A., Loredo, J., DeMiguel, E., and Charlesworth, S., 2003. Distribution of heavy metals in the street dusts and soils of an industrial city in Northern Spain. Archives of Environmental Contamination and Toxicology 44 (2): 160-170.

Orroño, D.I. and Lavado, R.S., 2009. Distribution of Extractable Heavy Metals in Different Soil Fractions. Chemical Speciation and Bioavailability 21: 193-198.

Probert, M.E., B.A. Keating, J.P. Thompson, and W.J. Parton. 1995. Modelling water, nitrogen, and crop yield for a longterm fallow management experiment. Australian Journal of Experimental Agriculture 35:941-950.

Ríos-Bustamante, A. 1992. Mexican Los Ángeles: A narrative and pictoral history, nuestra historia series Encino: Floricanto Press. pp:50–53.

Säumel, I., Kotsyuk, I., Hölscher, M., Lenkereit, C., Weber, F., and Kowarik, I. How healthy is urban horticulture in high traffic areas? Trace metal concentrations in vegetable crops from plantings within inner city neighbourhoods in Berlin, Germany. Environmental Pollution 165: 124-132.

Sauvé, S., Turmel, S.M., Roy, A.G., and Courchesne, F. 2003. Solid solution partitioning of Cd, Cu, Ni, Pb, and Zn in the organic horizons of a forest soil. Environmental Science and Technology 37:5191–5196.

Sipter, E., Rozsa, E., Gruiz, K., Tatrai, E., and Morvai, V., 2008. Site-specific risk assessment in contaminated vegetable gardens. Chemosphere 71: 1301-1307.

Schwarz, K., Pickett, S.T.A., Lathrop, R.G., Weathers, K.C., Pouyat, R.V., and Cadenasso, M.L. 2012. The effects of the urban built environment on the spatial distribution of lead in residential soils. Environmental Pollution 163: 32-39

Stilwell, D.R., Musante, C.L., and Sawhney, B.L. 2006. Effects of coating CCA pressure treated wood on arsenic levels in plants and soil. Frontiers in Plant Science 56(2): 2-6.

Stillwell, D.R., Rathier, T.M., and Musante, C.L. 2008. Comparison of Heavy Metals in Community Garden Produce versus Store-Bought Produce. The Connecticut Agricultural Experiment Station, Bulletin 1020.

Sun, Y., Wu, Q., Lee, C.C.C., Li, B., and Long, X. 2014. Cadmium Sorption Characteristics of Soil Amendments and its Relationship with the Cadmium Uptake by Hyperaccumulator and Normal Plants in Amended Soils. International Journal of Phytoremediation 16(5): 496-508.

Surls, R. and Gerber, J. 2010. “Los Angeles: A History of Agricultural Abundance”

Online available at:
http://ucanr.edu/blogs/blogcore/postdetail.cfm?postnum=3050

Thanabalasingam, P. and Pickering, W.F., 1986. Arsenic sorption by humic acids. Environmental Pollution 12: 233–246.

Virtanen, S., Vaaramaa, K., and Lehto, J. 2013. Fractionation of U, Th, Ra and Pb from boreal forest soils by sequential extractions. Applied Geochemistry 38: 1-9.

Wang, J., Ren, H., Liu, J., Yu, J., and Zhang, X. 2006. Distribution of lead in urban soil and its potential risk in Shenyang City, China. Chinese Geographical Science 16 (2): 127-132.

Wenzel, W. 2013. Arsenic. In *Heavy Metals in Soils: Trace Metals and Metalloids in Soils and their Bioavailability (third edition)*(Alloway, B.J. ed.). Environmental Pollution 22.

Wilson, S.C., Lockwood, P.V., Ashley, P.M., and Tighe, M. 2010. The chemistry and behaviour of antimony in the soil environment with comparisons to arsenic: A critical review. Environmental Pollution 158: 1169–1181.

Wu, J., Edwards, R., He, X., Liu, Z., and Kleinman, M. 2010. Spatial analysis of bioavailable soil lead concentrations in Los Angeles, California. Environmental Research 110: 309–317.

Yesilonis, I.D., James, B.R., Pouyat, R.V., Momen, B., 2007. Lead forms in urban turfgrass and forest soils as related to organic matter content and pH. Environmental Monitoring and Assessment 146(1-3): 1-17.

Yesilonis, I., Pouyat, R., and Neerchal, N.K., 2008. Spatial distribution of metals in soils in Baltimore, Maryland: role of native parent material, proximity to major roads, housing age and screening guidelines. Environmental Pollution 156 (3): 723-731.

Zhang, Q., Li, J.M., Xu, M.G., Song, Z.G., and Zhou, S.W. 2006. Effects of amendments on bioavailability of cadmium and zinc in compound contaminated red soil. Journal of Agro-Environmental Science (in Chinese) 25:861–865.

Tables

Table 4.1: Spearman's correlation coefficient matrix and significance for relationships between metal concentrations.

		Cd	As	Pb
Cd	Correlation Coefficient	-	.464**	.541**
As	Correlation Coefficient	.464**	-	.338**
Pb	Correlation Coefficient	.541**	.338**	-

**p<0.001

Table 4.2: Percent of overall metals found in each sequential fraction. Percentage was calculated from totals of each extraction divided by the sum of the overall concentration found by ALS

Metal	Exchangeable%	Reducible%	Organic%	Residual%
Pb	21%	42%	3%	35%
As	25%	N/A	4%	69%
Cd	69%	30%	15%	4%

Figures

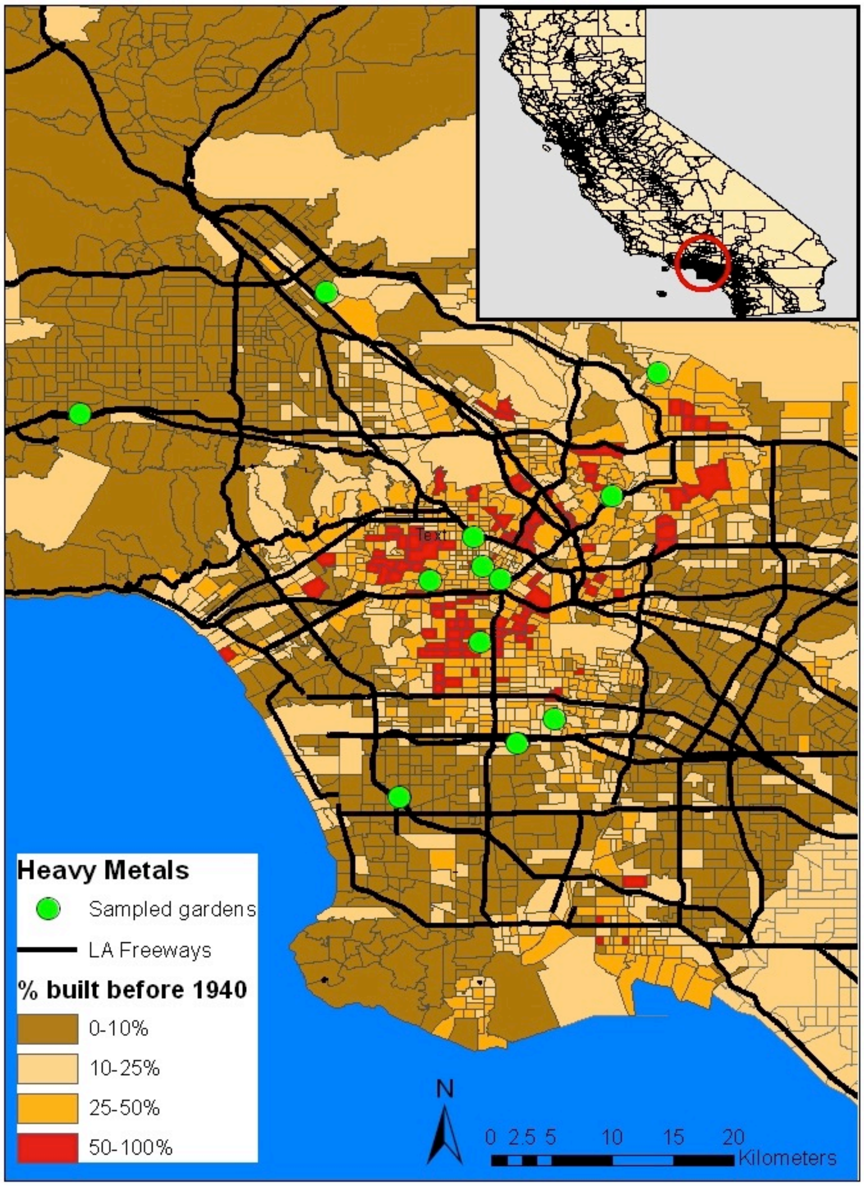

Figure 4.1: Map of Los Angeles County showing census tract boundaries (grey lines) and percent of structures built before 1940. Brown and beige have few older structures, orange and red have 25% or more. Black lines indicate the location of major freeways in LA. Green dots indicate community gardens where soil samples were collected.

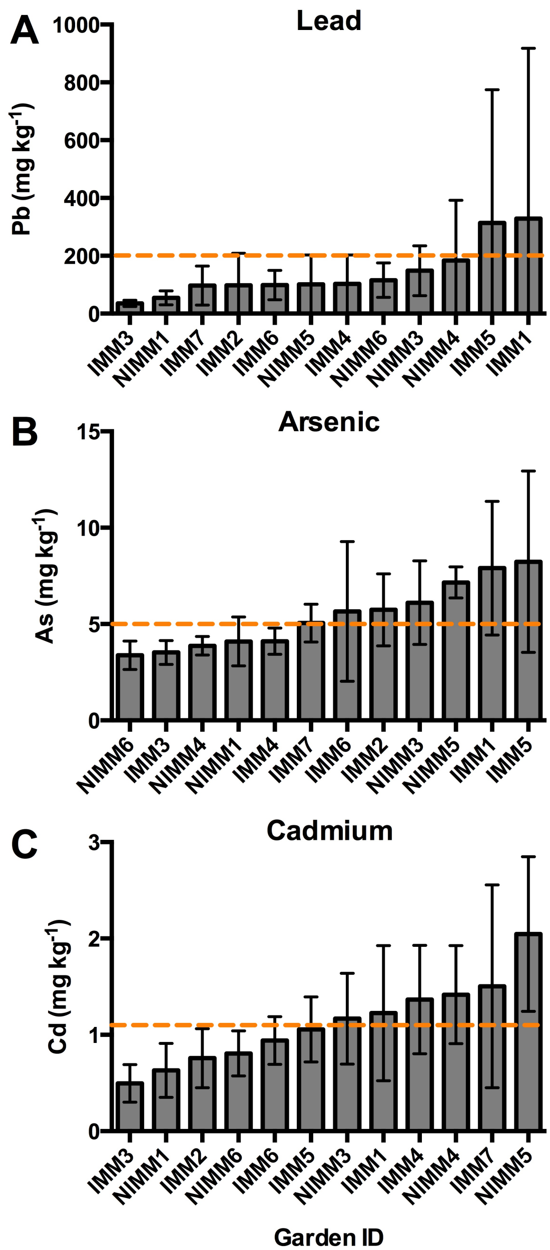

Figure 4.2: Mean garden scale soil concentrations for each of the three investigated metals. (A) Lead (Pb), (B) Arsenic (As), (C) Cadmium (Cd). Error bars represent standard error. The dotted lines on each graph represent background or safety levels for each metal. Pb: 200 mg/kg, the safety level in CA for where children play. As: 5 ppm, average background level for CA soils. Cd: 1.1 average background level in CA soils.

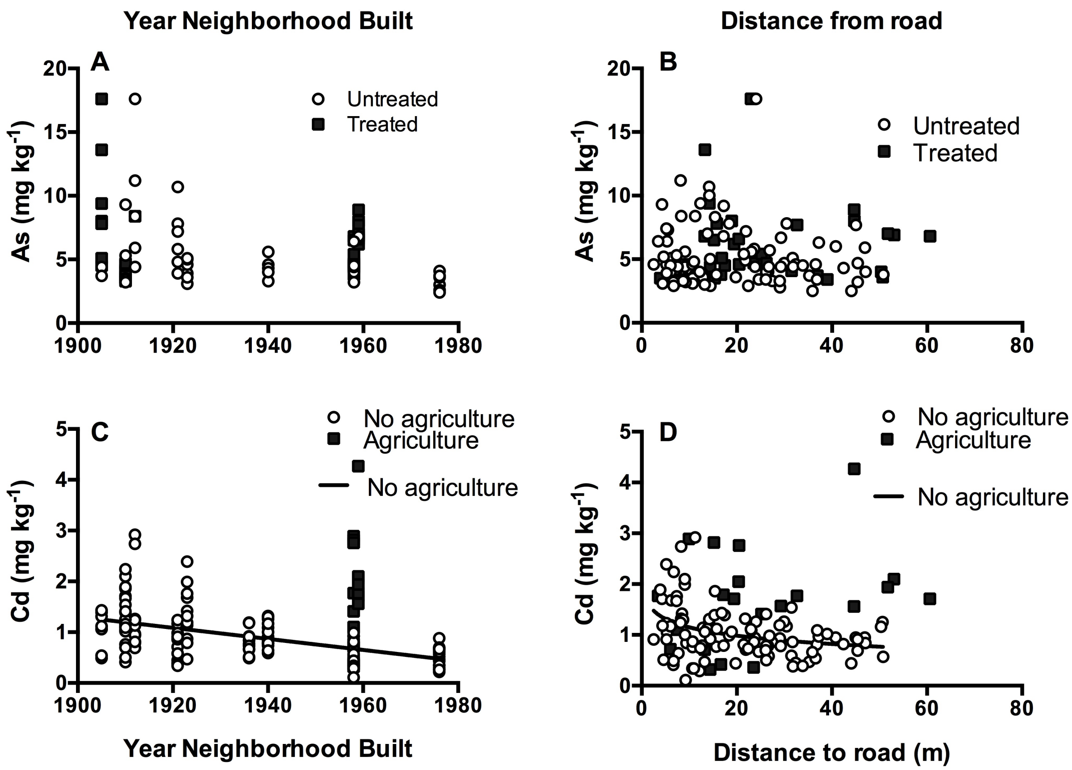

Figure 4.3: Effect of treated wood on year neighborhood build and distance from road patterns for Arsenic (A,B) and effect of agricultural land use history on Cd patterns (C, D). Lines represent significant relationships of Cd with neighborhood age ($p<0.05$, $r^2=0.259$), and distance from road ($p<0.05$, $r^2=0.129$) in non-agricultural locations.

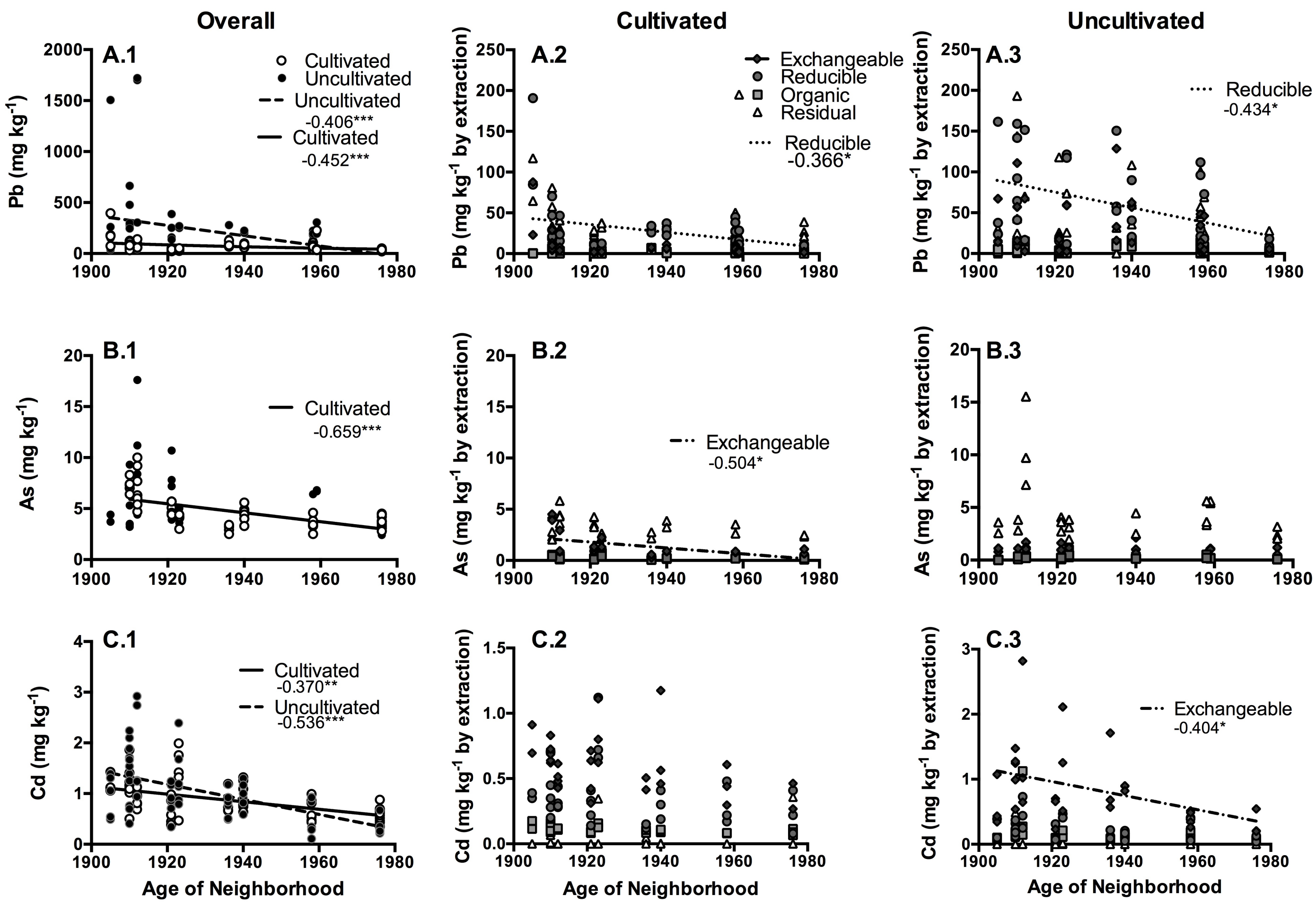

Figure 4.4: Curve fitted relationships between Pb (A.1-A.3), As (B.1-B.3), Cd (C.1-C.3) and age of neighborhood. Column 2 is sequential extraction of cultivated soils, Column 3 is uncultivated soils. Lines represent significant Spearman correlations curve fitted with AIC. Data for each significant relationship is included on each graph. $^*p<0.05$, $^{**}p<0.01$, $^{***}p<0.001$

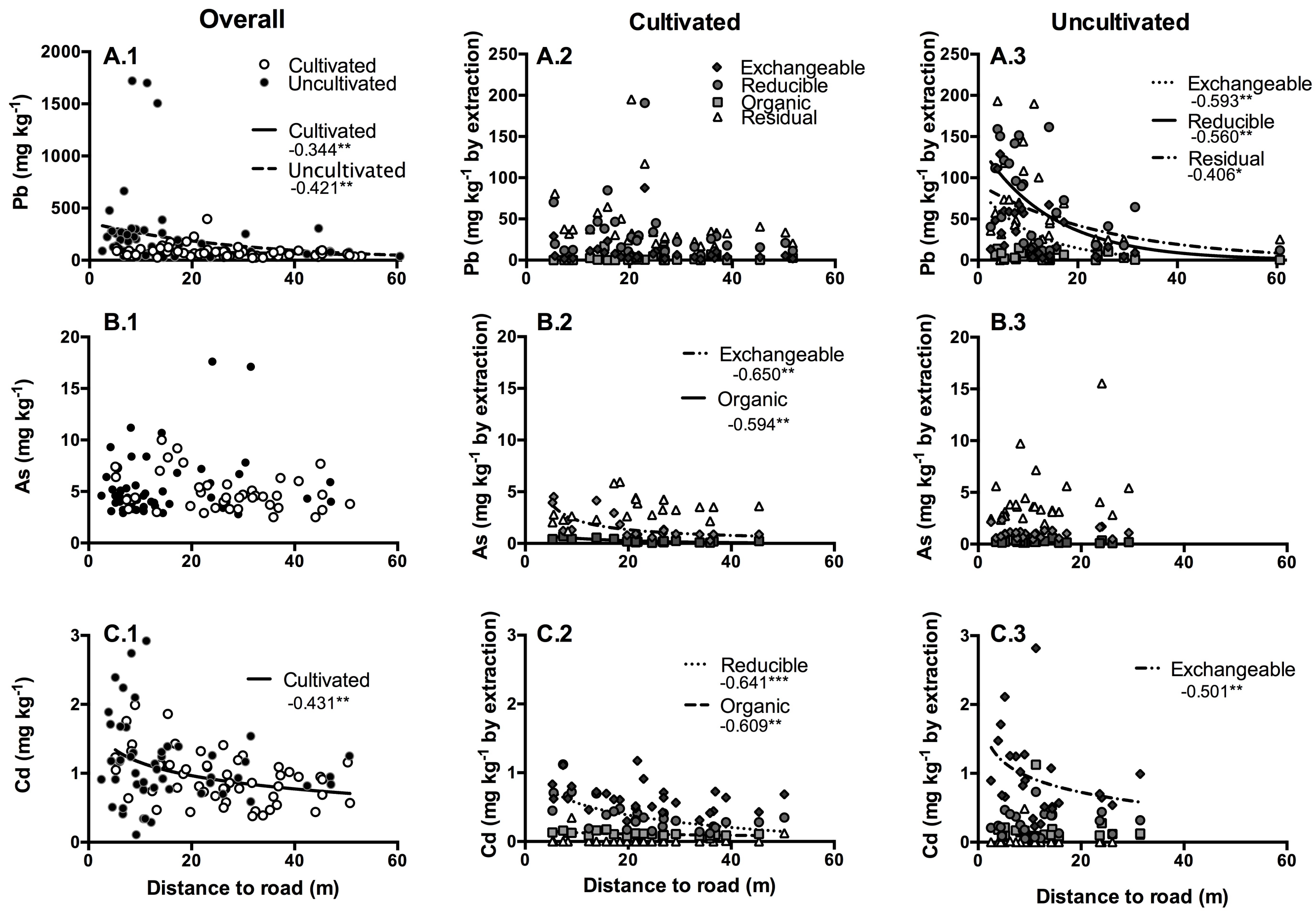

Figure 4.5: Curve fitted relationships between Pb (A.1-A.3), As (B.1-B.3), Cd (C.1-C.3) and distance from road. Column 2 is sequential extraction of cultivated soils, Column 3 is uncultivated soils. Lines represent significant Spearman correlations curve fitted with AIC. Data for each significant relationship is included on each graph. *p<0.05, **p<0.01, ***p<0.001

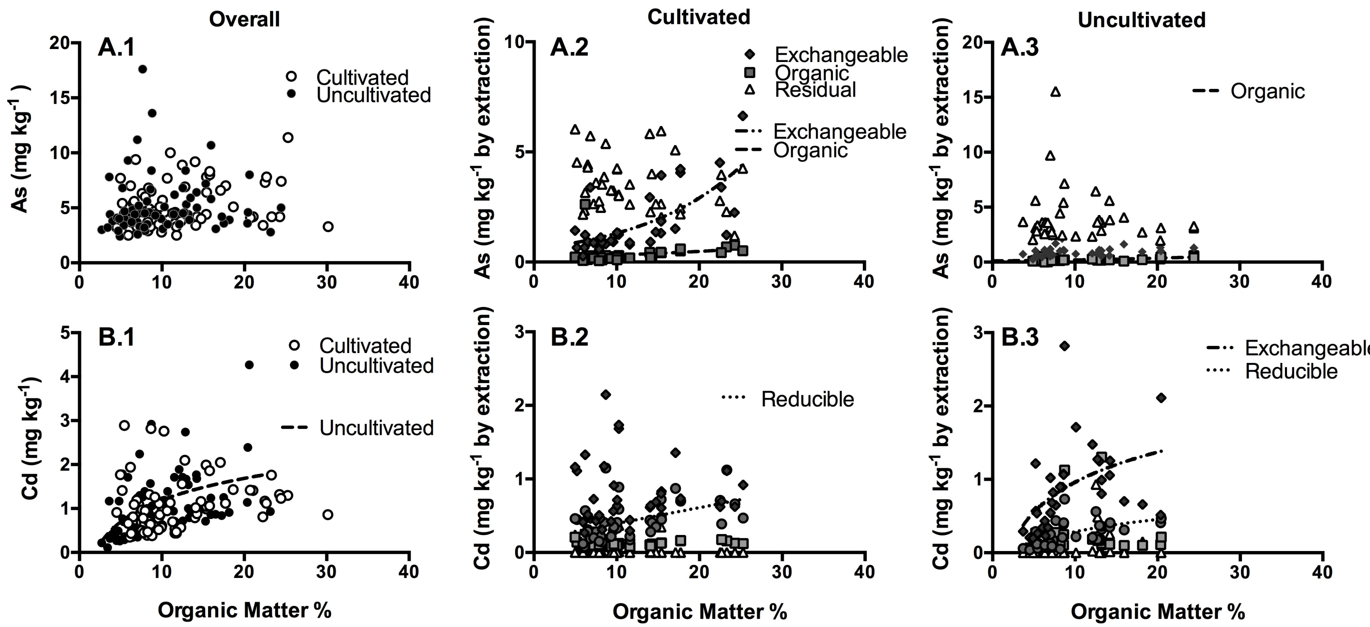

Figure 4.6: Curve fitted relationships between As (A.1-A.3), Cd (B.1-B.3) and OM content. Column 2 is sequential extraction of cultivated soils, Column 3 is uncultivated soils. Lines represent significant Spearman correlations curve fitted with AIC. Data for each significant relationship is included on each graph. *p<0.05, **p<0.01, ***p<0.001

Conclusion

The research presented in this dissertation was based on comprehensive crop biodiversity surveys conducted in community gardens of Los Angeles, CA and home gardens of Beijing, China, as well as soil surveys for heavy metal contamination in Los Angeles, CA. These studies addressed questions about driving mechanisms influencing the biodiversity, abundance, and direct ecosystem services (ES) and disservices in urban gardens across the world. In particular, these studies investigated the influence of income, culture, and neighborhood age on community garden composition and ES; distance from the city, income, and agricultural experience on home garden composition and ES; and age of neighborhood, distance from road, and garden management activities on heavy metal presence and speciation in community garden soils.

Through a three-year, comprehensive survey of community garden biodiversity and species uses, the second chapter asked, *What environmental and sociocultural variables influence diversity and abundance of community garden plants?* The results of this study identified influences of a hierarchy of need and cultural specificity in shaping plot, garden, and regional scale community garden species compositions and production of direct ecosystem services. Edible species are planted based on cultural background and demand for food production in immigrant gardens set in impoverished neighborhoods, while ornamentals proliferate in affluent neighborhoods for "luxury" aesthetic value. This shift from aesthetic to provisioning ES with reducing income is reflective of the hierarchy of need: with decreased financial resources, food becomes a priority. Management style of gardens affect planting patterns; species-area relationships exist in smaller gardens that do not communally share plots due to unmet demand for diversity and ES production with reduced space. Community gardens contribute to a bio-diverse urban ecosystem and positively contribute to food sovereignty through production of culturally relevant edible crops (Peña 2005; Lawson 2007).

The third chapter addresses home garden biodiversity and ES in five villages across an urbanizing gradient in Beijing, China, as many community gardeners come from countries with strong home garden traditions (Gottlieb 2006). It answers the question, *What variables influence diversity and abundance of home gardens in a developing country?* The results of this study are similar to the study of community gardens, in that it shows that home garden biodiversity shifts across urbanized regions in response to a hierarchy of need. Gardeners in suburban villages nearer to Beijing cultivate more aesthetically pleasing species, as they have higher income and more access to urban markets to buy food and ornamental plants. In more rural, exurban regions, gardeners cultivate more useful edible species, as they are isolated from the city and rely more heavily on agriculture to support their income. Edible and ornamental compositions were specific to each region, likely due to local agricultural traditions. The hierarchy of need also influenced species-area relationships, as low-income exurban villages were the only ones whose demand for edible diversity exceeded the garden space available. As large-scale agricultural production in China lacks vegetable biodiversity and local varieties (Yunlai and Fengying 2009), the diverse cropping techniques and food production in Chinese home gardens may preserve agricultural biodiversity and increase local food security.

Though both home and community gardens followed a hierarchy of need, the observed patterns were somewhat different. All individually-based community gardens surveyed in Los Angeles had strong species-area curves for provisioning species, while only exurban regions in Beijing had this pattern. Also, the observed hierarchy of need in LA was based on neighborhood income and culture, while home gardens in Beijing changed in both income and isolation from city resources, not in gardener cultural background. Fundamentally, these differences likely stem from the fact that urban home and community gardeners have different reasons for participating in gardens. In addition, these two agricultural spaces are structurally different. Villages in China have strong home garden traditions (Huai and Hamilton 2009) and the traditional *hutong* household structure is centered around a courtyard where people may choose to have a garden (Qi et al. 2008). Home gardeners have some control over how much of their courtyard is taken up by garden space and are less rigidly locked into available space than community gardeners are (Lawson and Drake 2013). This flexibility in home garden size may explain the lack of species-area curves outside of exurban villages where there is high demand for production of provisioning services. The decision to join a community garden is less casual than deciding to plant species in land that is owned by the gardener. Some gardens have waitlists of up to 10 years and participants must sign contracts after they join to keep the garden in cultivation during the duration of their stay, abide by the local rules, and pay a yearly fee of $10-100 per year (Lawson and Drake 2013; Vives 2009). This may indicate that participants in community gardens have a strong unmet need for ecosystem service production and are deliberately taking steps to remedy that situation. As noted in chapter 2, immigrant gardeners were likely to come from countries with strong home garden traditions (Peña 2006), like Chinese home gardeners. Though the described home and community gardens come from disparate urban regions and have innate differences in form and function, food production and hierarchy of need relationships are similar across socioeconomic gradients. The key to this similarity is a lack of certain resources, caused by low income or isolation, leading gardeners to plant species directly connected to their well-being. These results indicate the applicability of this framework to urban agricultural ecosystems across the world.

The reliance of both community and home gardens on provisioning edible service production to offset food security issues may also indicate vulnerability of gardeners to heavy metal exposure. The fourth chapter investigates the disservice of heavy metals in urban agricultural soils and their availability to planted crops. This study asks, *What factors influence the presence and bioavailability of heavy metals across community gardens in Los Angeles and how do these vary between metals of interest?* The main results indicate that legacies of land use, management, and buildings and air pollution are the main mechanisms of metal deposition. Increased reducible Pb in older neighborhoods indicates the presence of legacies of oxidized lead paint from older houses. Exchangeable Pb, As, and Cd all increased in proximity to roads due to vehicular pollution. This relationship suggests a clear management suggestion for placement of crops, as soils more than 20 meters from the road had drastically reduced bioavailable metals. Dilution of Pb and Cd in cultivated areas indicate that gardeners can also reduce levels of metal contamination through tillage and soil replacement. Organic matter (OM) and levels of pH are some of the most important soil properties under gardener control, as my results show that they interact in complex ways with As and Cd. While Cd becomes less bioavailable with increased OM due to an increase in adsorbing surfaces on OM particles, As becomes more bioavailable, due to reactions with humic acids in fertilizers (Wilson et al. 2010). Other contributing contamination factors included legacies of Cd with commercial agricultural background and As leaching from treated wood. Though Beijing home garden soils were not tested for contamination, other studies on elevated levels of heavy metals in Beijing due to air pollution indicate that home gardeners may also have increased exposure risks (Xia et al. 2011). These results can aid in risk assessment in urban residential soils (Schwarz et al. 2012), predicting metal accumulation hotspots, and remediating soil to reduce plant uptake and human exposure to accumulated metals.

In conclusion, this dissertation highlights the strong influence of gardener income, food security, cultural experience, and agricultural background on urban garden species composition. A hierarchy of need explains ecosystem service production in both home and community gardens across the world. Low-income gardeners plant and value species producing culturally relevant edible services, while higher income gardeners focus on aesthetics, as food production is less important to their well-being. These results indicate demand for urban planners to create and protect accessibility to urban gardens for residents in lower income neighborhoods. The observed focus on food production in gardens indicates high risk for exposure to soil contaminants from urban gardens. The collected results on metal presence and availability suggest clear management solutions for reduction of exposure to metals in urban gardens. These include removal of treated wood, moderation of pH and OM soil levels, cultivating soils 20 meters or more from busy roads, and testing soils for contamination in older neighborhoods. These findings on biodiversity, gardener preferences, and the production of ecosystem services and disservices will allow for better evaluations of complex coupled natural and human urban systems and their effect on human health and well-being.

Works Cited

Gottlieb, R. 2006. Reinventing Los Angeles; Nature and Community in the Global City. The MIT Press. Cambridge, Massachusetts, USA.

Huai, H. and Hamilton A. 2009. Characteristics and functions of traditional homegardens: A review. Frontiers of Biology in China 47:151– 157.

Lawson L. 2007. Cultural geographies in practice: The South Central Farm: Dilemmas in practicing the public. Cultural Geographies 14: 611-616.

Lawson, L. and Drake, L. 2013. Community Garden Organization Survey, 2011-2012. Community Greening Review. 18:20-41

Peña, D. 2005. "Farmers Feeding Families: Agroecology in South Central Los Angeles" Lecture presented to the Environmental Science, Policy and Management Colloquium, October 10, 2005.

Peña, D. 2006. "Toward a critical political ecology of Latina/o urbanism". The Acequia Institute. Online available at: <http://www.acequiainstitute.org/researchreports.html>

Qi, L., Li, D., Ma, D., Zhang, J., Li, S., Chu, J., and Xu, R. 2008. Study on the mode and structure arrangement of courtyard planting in the rural area of Wuxi. Journal of Jiangsu Forestry Science and Technology 35(1) 21-25 (In Chinese).

Schwarz, K., Pickett, S.T.A., Lathrop, R.G., Weathers, K.C., Pouyat, R.V., and Cadenasso, M.L. 2012. The effects of the urban built environment on the spatial distribution of lead in residential soils. Environmental Pollution 163: 32-39

Vives, R.. Article in Los Angeles Times, April 13, 2009. "Backyards could become community gardens in Santa Monica." Online available at: http://articles.latimes.com/2009/apr/13/local/me-santamonica-garden13

Wilson, S.C., Lockwood, P.V., Ashley, P.M., and Tighe, M. 2010. The chemistry and behaviour of antimony in the soil environment with comparisons to arsenic: A critical review. Environmental Pollution 158: 1169–1181.

Xia, X., Chen, X., Liu, R., and Liu, H. 2011. Heavy metals in urban soils with various types of land use in Beijing, China. Journal of Hazardous Materials 186: 2043-2050.

Yunlai, X. and Fengying, N. 2009 A report on the status of China's food security. China Agricultural Science and Technology Press, Beijing, China.