Urban Ecology for Conservation:

Evaluating Social and Ecological Drivers of Biodiversity Change Over Time

by

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ABSTRACT

Global biodiversity is threatened by anthropogenic impacts, as the global population becomes increasingly urbanized. Conservation researchers and practitioners increasingly recognize the potential of cities to support biodiversity and foster human-nature interactions. However, further understanding of social and ecological mechanisms driving change in urban biodiversity over time is needed. In this dissertation, I first synthesized evidence for the urban homogenization hypothesis, which proposes that cities are more similar across space and time than are the natural communities they replace. I found that approaches to testing urban homogenization varied widely, but there is evidence for convergence at regional spatial scales and for some taxa. This work revealed a lack of long-term urban studies, as well as support for social and ecological mechanisms driving homogenization.

Building from this systematic literature review, I tested the effects of a long-term nutrient enrichment experiment in urban and near-urban desert preserves to evaluate indirect urban impacts on natural plant communities over time. Urban preserves and nitrogen-fertilized plots supported fewer annual wildflower species, limiting their effectiveness for biodiversity conservation and nature provisioning for urban residents.

Finally, I conducted research on residential yards in Phoenix, Arizona, to explore the effects of individual management behavior on urban plant community dynamics. Using a front yard vegetation survey repeated at three time points and a paired social survey, I asked, to what extent are yard plant communities dynamic over time, and how do attitudes and parcel characteristics affect native plant landscaping? Front yard woody plant communities experienced high turnover on a decadal scale, indicating that these managed communities are dynamic and capable of change for conservation benefit. Residents held positive attitudes toward native plants, but cultivated few in their yards. Priorities such as desired functional traits, attitudes toward native plants, and household income predicted native plant abundance, while knowledge of native plants did not.

This body of work contributes to the growing understanding of how urban ecosystems change over time in response to local- and city-scale impacts, demonstrating opportunities to engage urban residents and land managers in local conservation action to improve the value of cities for people and biodiversity.

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INTRODUCTION

Cities cover less than 1% of land area globally, but are home to over half of the global population (Schneider et al. 2009, United Nations 2014). Despite their small land area, cities have disproportionate impacts on global carbon, nitrogen, and hydrologic cycles via both direct and indirect impacts (Vitousek et al. 1997, Grimm et al. 2008). Additionally, anthropogenic impacts threaten global biodiversity (Sala et al. 2000, Butchart et al. 2010, Newbold et al. 2015, Dornelas et al. 2019). As biodiversity increases the stability of ecosystems and their ability to respond to novel and changing conditions, the ongoing biodiversity crisis jeopardizes ecosystem function and resilience, particularly in urban and surrounding areas (Chapin et al. 2000, Cardinale et al. 2012).

With the growing urban population and the challenge of global biodiversity loss, cities have potential to provide conservation value. Cities can support surprisingly high biodiversity, including native and threatened species (Alvey 2006, McKinney 2008, Ives et al. 2016, Soanes et al. 2019). However, urban areas often promote generalist and non-native species while selecting against more specialized natives (McKinney and Lockwood 1999, Cadotte et al. 2017). Alterations to the abiotic environment and selection for a subset of species can result in urban homogenization, or greater similarity among cities than among natural environments (McKinney 2006). Changes in management action can increase biodiversity and promote native species in order to maintain local identity (Dearborn and Kark 2010, Aronson et al. 2017).

Urban biodiversity can be particularly important for urban residents, providing services such as psychological and mental health benefits, sense of place, and increased connection with nature (Fuller et al. 2007, Dearborn and Kark 2010, Hausmann et al. 2016). Urbanized populations can suffer from an "extinction of experience," or lost opportunities to experience and engage with the natural environment (Soga and Gaston 2016, Cox et al. 2017). This loss of connection has potentially wide-ranging impacts, as sense of connection with nature can predict conservation attitudes and behavior (Turner et al. 2004, Mackay and Schmitt 2019). Thus, increasing the exposure of urban residents to diverse natural landscapes is an important goal for conservation beyond direct ecological benefits.

To realize the potential contribution of cities to conservation strategies, it is necessary to understand how direct and indirect human actions shape the urban environment. Cities contain landscapes with a range of management intensities and types controlled by different kinds of managers (Larson et al. 2010, Sisser et al. 2016). These different landscapes can be evaluated to compare drivers of human actions and environmental outcomes. For example, residential landscapes are emergent products of management by many individual decision-makers, each of whom makes choices about what to do in their own yard. Their decisions are guided by personal, social, and contextual priorities and constraints, which impact management behavior and ecological function (Cook et al. 2012). Understanding how and why people change landscapes is essential for understanding the structure and function of Earth's many managed ecosystems.

In addition to direct management action, indirect effects of human activities extend outside of the metropolitan area and create an extensive human footprint on the landscape (Gregg et al. 2003, Erisman et al. 2013). While natural preserves in an urban context may not experience direct management, indirect effects can lead to decreased biodiversity and altered ecosystem function (Rogers et al. 2009). However, the mechanisms by which biodiversity is lost are often unclear and are therefore difficult to address with management action.

Current ecological research rarely evaluates temporal dynamics of urban landscapes, limiting understanding of how these ecosystems change and develop (Ramalho and Hobbs 2012). As in natural ecosystems, long-term studies are needed in order to disentangle responses to multiple drivers. Mechanistic research over time is particularly valuable in identifying opportunities for conservation and avoiding unintended consequences of interventions (Pataki 2015).

SCOPE AND STRUCTURE

In this dissertation, I explore the effects of direct and indirect urban influences on urban biodiversity, including through active management decisions. I aim to provide insight into actions that can achieve positive outcomes for biodiversity conservation in urban areas. I ask four questions related to the direct and indirect effects of cities and human management on urban biodiversity.

Chapter 2: How has urban homogenization been studied and described, and in what contexts does homogenization occur?

Chapter 3: How do long-term nutrient enrichment, water availability, and proximity to the urban environment interact to shape Sonoran Desert winter annual plant communities?

Chapter 4: To what extent are residential yard plant communities dynamic through time, and how rapidly do changes occur?

Chapter 5: How do resident attitudes, priorities, personal characteristics, and parcel structure predict native plant landscaping?

I address these questions using an interdisciplinary approach with multiple methods. In chapter 2, I conduct a systematic literature review to evaluate the multiple dimensions of urban homogenization research. Chapter 3 presents the results of a long-term experiment conducted as part of the Central Arizona-Phoenix Long Term Ecological Research program. In this chapter, I describe and quantify the effects of nutrient enrichment, interannual precipitation variability, and urbanization on winter annual plant diversity and community composition. In chapters 4 and 5, I use a repeated vegetation survey of Phoenix, Arizona, residential yards in combination with a resident social survey to characterize managed urban vegetation change over time and social-ecological drivers of landscape choices. Chapter 6 reviews the main findings from this work and evaluates next steps and implications for conservation management. Together, this body of work develops insight into the drivers of urban biodiversity change over time and provides guidance for urban residents and biodiversity conservation efforts.

REFERENCES

Alvey, A. A. 2006. Promoting and preserving biodiversity in the urban forest. Urban Forestry & Urban Greening 5:195–201.

Aronson, M. F. J., C. A. Lepczyk, K. L. Evans, M. A. Goddard, S. B. Lerman, J. S. MacIvor, C. H. Nilon, and T. Vargo. 2017. Biodiversity in the city: key challenges for urban green space management. Frontiers in Ecology and the Environment 15:189–196.

Butchart, S. H. M., M. Walpole, B. Collen, A. van Strien, J. P. W. Scharlemann, R. E. A. Almond, J. E. M. Baillie, B. Bomhard, C. Brown, J. Bruno, K. E. Carpenter, G. M. Carr, J. Chanson, A. M. Chenery, J. Csirke, N. C. Davidson, F. Dentener, M. Foster, A. Galli, J. N. Galloway, P. Genovesi, R. D. Gregory, M. Hockings, V. Kapos, J.-F. Lamarque, F. Leverington, J. Loh, M. A. McGeoch, L. McRae, A. Minasyan, M. Hernández Morcillo, T. E. E. Oldfield, D. Pauly, S. Quader, C. Revenga, J. R. Sauer, B. Skolnik, D. Spear, D. Stanwell-Smith, S. N. Stuart, A. Symes, M. Tierney, T. D. Tyrrell, J.-C. Vié, and R. Watson. 2010. Global biodiversity: indicators of recent declines. Science 328:1164–1168.

Cadotte, M. W., S. L. E. Yasui, S. Livingstone, and J. S. MacIvor. 2017. Are urban systems beneficial, detrimental, or indifferent for biological invasion? Biological Invasions 19:3489–3503.

Cardinale, B. J., J. E. Duffy, A. Gonzalez, D. U. Hooper, C. Perrings, P. Venail, A. Narwani, G. M. Mace, D. Tilman, D. A. Wardle, A. P. Kinzig, G. C. Daily, M. Loreau, J. B. Grace, A. Larigauderie, D. S. Srivastava, and S. Naeem. 2012. Biodiversity loss and its impact on humanity. Nature 486:59–67.

Chapin, F. S. III, E. S. Zavaleta, V. T. Eviner, R. L. Naylor, P. M. Vitousek, H. L. Reynolds, D. U. Hooper, S. Lavorel, O. E. Sala, S. E. Hobbie, M. C. Mack, and S. Díaz. 2000. Consequences of changing biodiversity. Nature 405:234–242.

Cook, E. M., S. J. Hall, and K. L. Larson. 2012. Residential landscapes as social-ecological systems: a synthesis of multi-scalar interactions between people and their home environment. Urban Ecosystems 15:19–52.

Cox, D. T. C., H. L. Hudson, D. F. Shanahan, R. A. Fuller, and K. J. Gaston. 2017. The rarity of direct experiences of nature in an urban population. Landscape and Urban Planning 160:79–84.

Dearborn, D. C., and S. Kark. 2010. Motivations for conserving urban biodiversity. Conservation Biology 24:432–440.

Dornelas, M., N. J. Gotelli, H. Shimadzu, F. Moyes, A. E. Magurran, and B. J. McGill. 2019. A balance of winners and losers in the Anthropocene. Ecology Letters 22: 847–854.

Erisman, J. W., J. N. Galloway, S. Seitzinger, A. Bleeker, N. B. Dise, A. M. R. Petrescu, A. M. Leach, and W. de Vries. 2013. Consequences of human modification of the global nitrogen cycle. Philosophical Transactions of the Royal Society B 368:20130116.

Fuller, R. A., K. N. Irvine, P. Devine-Wright, P. H. Warren, and K. J. Gaston. 2007. Psychological benefits of greenspace increase with biodiversity. Biology Letters 3:390–394.

Gregg, J. W., C. G. Jones, and T. E. Dawson. 2003. Urbanization effects on tree growth in the vicinity of New York City. Letters to Nature 424:183–187.

Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai, and J. M. Briggs. 2008. Global change and the ecology of cities. Science 319:756–760.

Hausmann, A., R. Slotow, J. K. Burns, and E. Di Minin. 2016. The ecosystem service of sense of place: benefits for human well-being and biodiversity conservation. Environmental Conservation 43:117–127.

Ives, C. D., P. E. Lentini, C. G. Threlfall, K. Ikin, D. F. Shanahan, G. E. Garrard, S. A. Bekessy, R. A. Fuller, L. Mumaw, L. Rayner, R. Rowe, L. E. Valentine, and D. Kendal. 2016. Cities are hotspots for threatened species. Global Ecology and Biogeography 25:117–126.

Larson, K. L., E. Cook, C. Strawhacker, and S. J. Hall. 2010. The influence of diverse values, ecological structure, and geographic context on residents' multifaceted landscaping decisions. Human Ecology 38:747–761.

Mackay, C. M. L., and M. T. Schmitt. 2019. Do people who feel connected to nature do more to protect it? A meta-analysis. Journal of Environmental Psychology 65:101323.

McKinney, M. L. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127:247–260.

McKinney, M. L. 2008. Effects of urbanization on species richness: a review of plants and animals. Urban Ecosystems 11:161–176.

McKinney, M. L., and J. L. Lockwood. 1999. Biotic homogenization: a few winners replacing many losers in the next mass extinction. Trends in Ecology & Evolution 14:450–453.

Newbold, T., L. N. Hudson, S. L. L. Hill, S. Contu, I. Lysenko, R. A. Senior, L. Börger, D. J. Bennett, A. Choimes, B. Collen, J. Day, A. De Palma, S. Díaz, S. Echeverria-Londoño, M. J. Edgar, A. Feldman, M. Garon, M. L. K. Harrison, T. Alhusseini, D. J. Ingram, Y. Itescu, J. Kattge, V. Kemp, L. Kirkpatrick, M. Kleyer, D. Laginha Pinto Correia, C. D. Martin, S. Meiri, M. Novosolov, Y. Pan, H. R. P. Phillips, D. W. Purves, A. Robinson, J. Simpson, S. L. Tuck, E. Weiher, H. J. White, R. M. Ewers, G. M. Mace, J. P. W. Scharlemann, and A. Purvis. 2015. Global effects of land use on local terrestrial biodiversity. Nature 520:45–50.

Pataki, D. E. 2015. Grand challenges in urban ecology. Frontiers in Ecology and Evolution 3:57.

Ramalho, C. E., and R. J. Hobbs. 2012. Time for a change: dynamic urban ecology. Trends in Ecology & Evolution 27:179–188.

Rogers, D. A., T. P. Rooney, T. J. Hawbaker, V. C. Radeloff, and D. M. Waller. 2009. Paying the extinction debt in southern Wisconsin forest understories. Conservation Biology 23:1497–1506.

Sala, O. E., F. S. III Chapin, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, M. Oesterheld, N. L. Poff, M. T. Skykes, B. H. Walker, M. Walker, and D. H. Wall. 2000. Global biodiversity scenarios for the year 2100. Science 287:1770–1774.

Schneider, A., M. A. Friedl, and D. Potere. 2009. A new map of global urban extent from MODIS satellite data. Environmental Research Letters 4:044003.

Sisser, J. M., K. C. Nelson, K. L. Larson, L. A. Ogden, C. Polsky, and R. Roy Chowdhury. 2016. Lawn enforcement: how municipal policies and neighborhood norms influence homeowner residential landscape management. Landscape and Urban Planning 150:16–25.

CHAPTER 2

CONVERGENCE OF WHAT? A REVIEW OF THE EVIDENCE FOR URBAN HOMOGENIZATION

Abstract Cities are unique biomes that appear to share similar characteristics in form and function across the planet. As cities continue to expand globally, reduced uniqueness of local environments can threaten local biodiversity and human connection to nature. As first defined in 2006, the urban homogenization hypothesis explores the increasing biotic similarity of cities through replacement of native species with cosmopolitan non-natives. A growing body of literature has investigated this hypothesis, which has since extended to explorations of abiotic convergence and increased biotic similarity via other mechanisms beyond species introductions. Widely varying definitions, approaches, and spatial scales used to test the urban homogenization hypothesis impede clear understanding of the empirical support for this pattern of convergence and the mechanisms underlying it. We conducted a systematic review of the peer-reviewed literature to synthesize existing research on the topic of urban homogenization. Specifically, we asked how homogenization is defined and measured, for which taxa, characteristics, and spatial scales it has support, and what mechanisms are proposed as drivers of this process. Across 57 studies from 2006 to 2019, the evidence for homogenization was much stronger for some taxa (e.g., birds and plants) than others (e.g., beetles). Most studies tested taxonomic homogenization, with relatively few investigations of abiotic, phylogenetic, or functional homogenization. Non-native species had a homogenizing effect in urban ecosystems overall, although this effect may depend on time since introduction. Although the literature reveals patterns of reduced urban

distinctiveness at local to regional scales, to date, there is limited support for global-scale ecological homogenization of urban ecosystems. Consistent use of unified definitions and theory of urban homogenization will further improve our understanding of current and future ecology of urban and urbanizing landscapes.

INTRODUCTION

Cities are expanding globally, with widespread effects on the abiotic environment and biotic communities (Grimm et al. 2008). One hypothesized change resulting from urbanization is the loss of local distinctiveness leading to increased similarity among cities in comparison to natural environments, or "urban homogenization" (McKinney and Lockwood 1999, McKinney 2006). Urban homogenization threatens the local distinctiveness of cities with important implications for ecosystem services and urban resilience (Cardinale et al. 2012). Sense of place and local character are valuable for human health and wellbeing and depend on the uniqueness of individual cities (Hausmann et al. 2016). Further, efforts to preserve the unique environmental and biotic characteristics of cities based on their geographic location may facilitate conservation efforts by decreasing the proliferation of common urban-adapted species. Thus, understanding the extent and drivers of urban homogenization serves both human and conservation goals.

The study of urban homogenization builds on early work describing biotic homogenization as the process of increasing community similarity through exotic species introductions and loss of natives (McKinney and Lockwood 1999). This seminal paper was a call to action for biodiversity conservation, raising concern that human activity would lead to a globally homogenized biosphere with limited diversity in the absence of intervention. Subsequently, two distinct bodies of literature developed. Research on biotic homogenization across numerous ecosystems has explored whether invasion and extinction are leading to increasing biotic similarity over space and time, without reference to urbanization in particular (Olden 2006, Rooney et al. 2007, Olden et al. 2018). Another line of research focuses on urbanization as a primary driver of homogenization, and has grown rapidly following the publication of a 2006 special issue of Biological Conservation (McKinney 2006). Recently, the urban-specific homogenization literature has begun to investigate abiotic similarity in addition to patterns of biotic community change, sometimes referred to as "urban ecological convergence" (Groffman et al. 2014, 2017).

While biotic homogenization refers to increased similarity through processes that increase non-native species and decrease native species, urban homogenization is most often tested as a pattern with multiple possible driving mechanisms. Urbanization may create similar environmental conditions through direct change (e.g., to land cover) and as an indirect result of human activities (e.g., altered biogeochemical cycles; Kaye et al. 2006, Groffman et al. 2014). Shared abiotic conditions may subsequently promote similar species while disadvantaging locally adapted natives, resulting in homogenization (McKinney 2006). Purposeful introduction and cultivation of the same species may also result in similar urban communities (Bradley et al. 2012, Padullés Cubino et al. 2019). Accidental introductions of invasive species through transportation of propagules and stowaways may also be more common in urban areas, resulting in spread of non-natives within and surrounding the city (von der Lippe and Kowarik 2008, Adler and Tanner

2013). These drivers of urban homogenization can be mitigated by different types of interventions to maintain local ecological uniqueness, but research to date often does not distinguish between drivers of increased similarity.

Many approaches have been used to test biotic homogenization, resulting in a lack of clarity around the drivers and implications of this process at the global scale (Olden et al. 2018). The urban homogenization literature, which encompasses a broader suite of subjects (both biotic and abiotic) and driving mechanisms, includes even greater variability in approaches and results. Some studies test urban homogenization as greater similarity between cities compared to between natural environments (e.g., Aronson et al. 2014, Steele et al. 2014), some as greater similarity among more urban compared to less urban land use types within the same city (e.g., Hartley et al. 2007, Rocha et al. 2016), and some as greater similarity among non-native species than among native species (Horsák et al. 2013, Garcillán et al. 2014). Many studies make comparisons at a single timepoint, but some test for increasing similarity over time (e.g., Marchetti et al. 2006, Rogers et al. 2009). Taken together, these diverse approaches present an opportunity to evaluate the underlying drivers promoting urban homogenization.

Using a systematic literature review, we assessed the evidence in support of urban homogenization and how these findings implicate different drivers of this pattern. Specifically, we considered how each study addressed the following aspects of urban homogenization.

1. Definition of homogenization and metric used (how is homogenization *measured*?)
2. Taxa and abiotic conditions studied (homogenization *of what*?)

3. Comparisons made (homogenization *relative to what*?)
4. Spatial extent (homogenization at what *scale*?)
5. Proposed pathways (by what *mechanisms* did homogenization occur?)

We hypothesized that homogenization of urban ecosystems is mostly promoted by direct management, leading to greater convergence of managed subjects, such as cultivated plants and land cover, and places that experience more active management, such as manicured parks relative to natural preserves. Due to the importance of management, we also predicted that homogenization would occur across spatial scales that experience similar types of management, such as cities with similar climates.

METHODS

In order to assess the state of urban homogenization research relative to our five questions, we conducted a systematic literature review following meta-analysis procedures outlined by Koricheva et al. (2013). We ran three searches in Web of Science[1] with three sets of search terms, where $ is any one or no characters and * is any number of characters. The first aimed to identify studies of urban homogenization or urban ecological convergence: (*urban* NOT disturban*) AND (homogen* OR heterogen* OR converg* OR diverg*) AND (ecolog* OR environment*). This search was limited to ecological or environmental studies. The word segment "disturban*" was excluded to prevent retrieval of non-urban studies of disturbance, as the wild characters around

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[1] Citation indexes: Science Citation Index Expanded (SCI-EXPANDED) --1900-present, Social Sciences Citation Index (SSCI) --1956-present, Arts & Humanities Citation Index (A&HCI) --1975-present, Emerging Sources Citation Index (ESCI) --2015-present

"urban" retrieved all studies with the word disturbance as well. While this limitation may have omitted some studies of disturbance and urban homogenization, it helped to maintain a narrower focus on urban homogenization processes rather than disturbance more generally.

The next set of search terms retrieved papers making multi-city comparisons, as one approach to urban homogenization tests for decreased variance across multiple cities: (cross$city OR multi$city OR “multiple cities” OR “multiple city”). While this approach included many studies not aimed at urban homogenization or ecological questions, it broadly addressed multi-city comparisons with a variety of frames. The final search targeted papers testing for beta diversity in the urban environment as one approach to increasing community similarity among urban landscapes using the search term (urban beta-diversity). This search retrieved only studies focused on biota, but complements the other searches by expanding the possible framings of urban homogenization included.

All searches encompassed the titles, keywords, and abstracts of journal articles and proceedings papers (reviews not included) published in English. We last ran these searches on May 2, 2019; papers published after this date are not included.

The initial search process yielded a total of 5,195 articles (Fig. 1). We followed an inclusion/exclusion sorting process to select only the most relevant papers to draw conclusions about urban homogenization. As a first step, we removed 1,272 articles published in journals which were clearly outside the scope of this research (e.g., focused on medicine and human health, transportation, or technology; Appendix A: Table S1). A research assistant reviewed these article titles and abstracts where necessary to confirm exclusion by journal. Next, two research assistants read titles and abstracts for the 3,923 remaining papers, 1,106 of which at least one reviewer believed might qualify for inclusion. We then reviewed these papers in full to determine the final set of papers for inclusion. All included studies met the following requirements.

1. Used original data (i.e., no reviews, conceptual papers, or modeling papers without new data).
2. Focused on non-human biota or abiotic conditions as the main subject (e.g., not human health, animal behavior, sustainability, risk exposure, design guidance).
3. Study sites included the urban environment (e.g., studies of only natural forests were excluded, but studies of forest patches with surrounding urbanization could be included if they met all other criteria).
4. Tested homogenization, as defined by the individual paper.
5. Included a hypothesis related to urban homogenization or convergence.

Additionally, we excluded papers that made comparisons along urban to rural gradients but did not define their questions and analyses in terms of urban homogenization or ecological convergence. For example, studies comparing species richness or soil characteristics in urban, suburban, and natural environments with no tests of within-group similarity were excluded, as they tested simply for differences among these environments without testing for decreased variability in more urbanized spaces.

64 articles met all criteria (Appendix A: Table S2). Of these 64 papers, seven pairs analyzed the same datasets and thus cannot be described as reaching independent conclusions: (Aronson et al. 2014, La Sorte et al. 2014a), (Blair 2004, Blair and Johnson 2008), (Horsák et al. 2013, 2016), (Knapp and Wittig 2012, Huwer and Wittig 2013), (La Sorte and McKinney 2006, La Sorte et al. 2007), (Lososová et al. 2012, 2016), and (La

Sorte et al. 2008, Ricotta et al. 2014). In these cases, either the first published or the more inclusive paper was included (more sites - La Sorte et al. 2007, more sites - Blair and Johnson 2008, first published - Lososová et al. 2012, first published - Horsák et al. 2013, more sites - Huwer and Wittig 2013, more taxa - Aronson et al. 2014, more sites - Ricotta et al. 2014).

For each paper in the final dataset (n = 57), we recorded the type of homogenization tested, the metric used, the taxa or abiotic condition studied, the urban and non-urban comparisons made, the location and spatial extent of the study, and the conclusions reached by the authors about whether urban homogenization was supported (Table 1). We used a vote counting approach to tally the conclusions of all studies (i.e., that urban homogenization had or had not occurred) grouped by research approach. Some studies found mixed evidence for and against urban homogenization, or for homogenization in part but not in full. These were classified as reaching mixed conclusions but found at least some support for urban homogenization.

FINDINGS

Of the 57 included studies, 27 (47%) found support for the urban homogenization hypothesis and 16 (28%) found mixed results with at least some evidence for homogenization (Appendix A: Table S2). Studies with mixed results either addressed multiple types of homogenization using multiple metrics (3 studies, see Question 1), considered multiple taxa, species groupings, or subjects (8 studies, see Question 2), compared multiple site types (3 studies, see Question 3), or addressed homogenization at multiple spatial scales (3 studies, see Question 4). 14 studies (25%) concluded that their findings did not support the urban homogenization hypothesis, either showing no convergence among urban compared to non-urban landscapes or finding evidence of differentiation with urbanization (i.e., urban landscapes were less homogeneous).

Question 1.1: How is homogenization described?

Urban homogenization was generally either defined by biotic homogenization via the increase of widespread, generally non-native species and decline of endemic, rare, or native species, or described in broader terms including biotic homogenization as one part of urban homogenization. While some studies focused descriptions of urban homogenization mostly on nativity, cosmopolitan species, or spread of generalists, it was more typical for studies to reference all three of these ideas as part of their introduction of the urban homogenization hypothesis. Variation in which species defined the pattern of urban homogenization highlights the difficulty of comprehensively testing and addressing the urban homogenization hypothesis as a single idea. Rather, this hypothesis is multi-faceted and different conclusions at least partly depend on the approach taken.

The similarity of abiotic features such as soil nutrient levels and surface water distribution are increasingly investigated as part of urban homogenization, but cannot be described with the same framing as community-based studies, that is, as increased community similarity. A small group of papers used the term "urban ecological convergence" as a way of capturing increased urban similarity not limited to biotic interchange. This term is more appropriate for abiotic features of the environment, as it is more distant from the implied biological mechanisms of biotic homogenization. Additionally, while biota may be characterized as moving and mixing - that is, homogenizing - these connotations are less suitable for characteristics of soils or water distribution, which can become more similar in urban spaces but do not mix in the same sense. While usage of the term urban ecological convergence remains limited, this term may be more appropriate in future research examining the increased similarity of cities through space and time. A restructuring of the terminology describing the urban homogenization hypothesis will allow greater clarity in future research directions.

Question 1.2: What types of homogenization?

Seven types of homogenization were tested in the included papers: taxonomic, phylogenetic, functional, structural, historical, temporal, and abiotic (Table 2). Taxonomic homogenization (i.e., increasing similarity in species composition) was by far the most frequently explored (91% of studies), while other types of homogenization were tested in less than 10% of studies.

Phylogenetic homogenization (i.e., greater evolutionary relatedness among urban communities) was supported for urban birds and bees (Morelli et al. 2016, Harrison et al. 2018). In residential yard plant communities, wild-growing non-natives increased phylogenetic homogenization while cultivated species tended to differentiate yard floras (Padullés Cubino et al. 2019), though cultivated pools were still more phylogenetically related than flora in natural communities (Pearse et al. 2018). The only study of structural homogenization found more similar tree density in residential yards across regions than in natural communities (Pearse et al. 2018).

Four studies considered functional homogenization (i.e., increasing similarity of species traits such as size and diet type). These results showed functional homogenization of bird and ant communities (Holway and Suarez 2006, Luck and Smallbone 2011), but differentiation of riparian forest plant communities (Brice et al. 2017). Desert freshwater fish communities experienced both taxonomic and functional homogenization over time, but homogenization was not related to the amount of urban land cover in the watershed (Pool and Olden 2012).

While studies of biotic homogenization typically compare communities over time to test for increasing similarity (Olden et al. 2018), only five studies (9%) compared current with historical data to test whether homogenization over time was greater in urban areas. Findings supported greater homogenization over time for plants and birds in urban environments (Rogers et al. 2009, Catterall et al. 2010, Huwer and Wittig 2013). However, studies of freshwater fish found community differentiation over time associated with urbanization (Marchetti et al. 2006) or homogenization over time, but driven more by dams than by urbanization (Pool and Olden 2012). While several studies call for more temporal analyses, most relied on a space-for-time approach to compare pre- and post- urbanization conditions. The few temporally explicit studies show that increased similarity through time is likely, but not well studied, and not necessarily driven by urbanization for all organisms.

Five studies tested temporal homogenization (i.e., decreased seasonal or interannual variability in urban environments). Four of these studies examined bird communities and found reduced seasonal and interannual variability in urban environments (La Sorte et al. 2014b, Leveau et al. 2015, Leveau and Leveau 2016, Kale et al. 2018). The fifth study found reduced temporal turnover of both butterflies and plants with increasing urbanization (Uchida et al. 2018). The available evidence suggests that seasonal and interannual variability is reduced in urban environments, though taxa other than birds are not well represented.

Homogenization of the abiotic environment was measured by four papers, which tested convergence of surface water (Steele et al. 2014), microclimate (Hall et al. 2016), and soil characteristics (Jenerette et al. 2006, Pouyat et al. 2015). Surface water distribution, air temperature, and humidity converged in cities relative to natural landscapes across the U.S. (Steele et al. 2014, Hall et al. 2016). Soil characteristics varied, with convergence of pH, organic carbon, and total nitrogen across three continents, but divergence in phosphorus and potassium concentrations (Pouyat et al. 2015). These mixed results were explained by the importance of management behaviors for soil carbon and nitrogen concentrations, while phosphorus and potassium are more influenced by parent material. In a single city, Jenerette et al. (2006) found reduced within-park variability in urban soils, but greater variability among urban parks than among natural preserves.

Question 1.3: How is homogenization measured?

Ecologists measure community similarity with many metrics, and ongoing disagreement over the best metrics also impacts urban homogenization research (Anderson et al. 2011, Olden et al. 2018). Almost none of the included papers used the same approach to testing urban homogenization, with variation either in the type of comparison made (e.g., landscapes and spatial scales compared, see Questions 3 and 4) or in the metric used (e.g., beta diversity, homogenization index). Additionally, abiotic convergence is not easily measured by community composition approaches, so most of these studies used coefficient of variation or standard deviation within urban and non-urban locations as metrics of homogenization, while few studies of biota used this approach. A recent review of studies of biotic homogenization noted that evaluating findings across studies using quantitative meta-analysis approaches would be unlikely to yield meaningful insight into the magnitude of effects due to the wide variation in metrics used (Olden et al. 2018). The urban homogenization literature is broader still and even more susceptible to this challenge.

Urban homogenization of biota was most often measured by comparing beta diversity (i.e., species turnover between communities), represented by Jaccard, Sørensen, or Bray-Curtis similarity. 21 studies (37%) compared average within-habitat beta diversity for urban and non-urban sites, 8 (14%) used cluster analyses to see if urban areas grouped apart from natural areas, and 4 (7%) compared distance to the group average as a measure of dispersion (Table 3). These approaches are all similar in that they compare overall composition among communities of different types to test whether urban communities have more in common than non-urban communities.

Partitioning beta diversity into turnover and nestedness components can reveal changes in species identities (turnover, often measured with the β_{sim} index) compared to changes in diversity (nestedness; Baselga 2010). 16 included studies (28%) used this approach, with three focused exclusively on the turnover component of beta diversity (La Sorte et al. 2007, 2014b, Pino et al. 2009) and one on only the nestedness component (Ulrich et al. 2007). Analyses of total and partitioned beta diversity did not always yield the same results. For example, when comparing similarity of native, extralimital native (species introduced from nearby sources), and exotic plant communities in the northeastern U.S. using the same dataset, one study using total beta diversity found the greatest similarity among extralimital natives, while another using only the turnover component of beta diversity found the greatest similarity among exotic species (La Sorte and McKinney 2006, La Sorte et al. 2007). The choice of metric reflects different types of homogenization, either due to overall increasing similarity among sites (total beta diversity), decreased diversity in urban areas (nestedness), or more similar groups of species regardless of diversity (turnover; Olden et al. 2018).

Another approach compared changes in community similarity over geographic distance (9 studies, 16%). These studies hypothesized that more geographically separated locations are naturally more dissimilar, but that similarity may change less with distance in urban areas than in non-urban areas. Some studies found no correlation between community similarity and geographic distance, indicating homogenization (Asmus and Rapson 2014), or found a smaller decrease in similarity over distance for urban compared to non-urban communities (Luck and Smallbone 2011, Murthy et al. 2016). Others compared change in similarity with distance for native and non-native species, testing homogenization by non-natives in particular (Lososová et al. 2012). However, most studies found no difference in the rate of change in similarity over distance for urban and non-urban communities (Table 3).

With these similarity-based approaches, researchers used a variety of metrics. The abundance-based Bray-Curtis index (13 studies, 23%) and presence-based Jaccard index (16 studies, 28%) and Sørensen index (12 studies, 21%) were most often used, with less frequent usage of the Morisita-Horn index (abundance-based, 3 studies, 5%) and Raup-Crick index (presence-based, 1 study, 2%). The metric chosen for a particular study may impact the findings: in one study, patterns of homogenization for tree communities were stronger when measured with abundance-based rather than presence/absence dissimilarity metrics (Yang et al. 2015). However, we found that 92% of studies using only the Jaccard index and 63% of studies using only the Sørensen index found evidence of homogenization, compared to 50% of studies using only the abundance-based Bray-Curtis index.

Four included studies (as well as two excluded for data overlap) tested homogenization via increasing non-native species relative to native species using the homogenization index (La Sorte and McKinney 2006, La Sorte et al. 2008, Lososová et al. 2012, Horsák et al. 2013, Garcillán et al. 2014, Padullés Cubino et al. 2019). The homogenization index is the difference between overall community similarity (typically measured by the Jaccard index) and community similarity of native species only, thus explicitly testing for homogenization by non-native species (Rahel 2000). This index would not capture homogenization of native species, that is, if urban communities all contained the same group of native species with a few different non-natives, but rather compares similarity among natives to similarity among non-natives. How broadly nativity is defined is of particular importance with this approach, as classifying a species that occurs in multiple sites as native to one but not another would change the way it impacts the homogenization index. All six of these papers found at least some evidence of homogenization, and all but one was focused on plants (see section 2).

Some studies classified species as cosmopolitan or endemic to create metrics of homogenization, such as percent native and percent cosmopolitan species (Deák et al. 2016) or relative abundance of endemic and cosmopolitan species (Scott 2006). In a related approach, Newbold et al. (2018) compared range sizes for urban and non-urban species globally and found that human-dominated land uses contained species with larger ranges (i.e., more cosmopolitan species). However, at a much smaller spatial scale comparing only cities in the northeastern U.S., La Sorte and McKinney (2006) found that native urban plants had the largest and least variable range sizes compared to extralimital natives and exotic species, suggesting scale-dependence of range size analyses.

Question 2: Homogenization of what?

Urban homogenization studies primarily addressed terrestrial communities and characteristics (49 studies, 86%), with only eight in aquatic, coastal, or wetland environments. In wetland areas, mammal communities closer to urban environments were more similar to one another than were mammal communities that were farther from urban areas, but there was no evidence of increased similarity of amphibians, reptiles, crayfish, mollusks, or aquatic insects (Johnson et al. 2013, Reichert et al. 2017). The two rocky intertidal studies shared some sites, and neither found evidence of homogenization in more urban locations (Oliveira et al. 2014, Bertocci et al. 2017). Aquatic studies of urban homogenization may be rare because these habitats are more impacted by direct changes to waterways (e.g., damming, channelization) than by indirect effects of urbanization (Marchetti et al. 2006, Scott 2006, Pool and Olden 2012). Consequently, studies of fish are much more commonly framed as biotic homogenization rather than urban homogenization (Olden et al. 2018).

Plants were the most commonly studied subject (22 papers, 39%, Fig. 2; see below). Birds were the next most frequent (14 papers, 25%), and homogenization of bird communities was overwhelmingly supported (but see Aronson et al. 2014). Twelve papers (21%) measured terrestrial invertebrates, with mixed findings (Appendix A: Table S3). None of the four studies of beetles found evidence of homogenization across various urban locations and with diverse approaches (Hartley et al. 2007, Ulrich et al. 2007, Magura et al. 2010, Knop 2016).

Nativity, cultivation, and homogenization of plant communities

Different approaches were taken to testing for floristic homogenization compared to homogenization of other organisms. Many studies focused on place of origin and duration of residence for plants, rather than just nativity. These studies investigated whether the homogenizing effects of recently introduced species differed from those that were more established. They hypothesized that recently arrived species would be patchily distributed and not yet present in their entire possible range (resulting in community differentiation across space) while established non-natives would be widespread (resulting in homogenization). Thus, these studies often compared natives, recently introduced species, and established non-natives to evaluate the homogenizing effects of different groups. In Europe, this translated to comparisons of natives, neophytes (species introduced after 1500), and archaeophytes (species introduced before 1500).

Results of studies in Europe showed strong support for differentiation by neophytes and homogenization by archaeophytes (La Sorte et al. 2007, 2008, 2014a, Knapp and Wittig 2012, Lososová et al. 2012, 2016, Huwer and Wittig 2013, Ricotta et al. 2014, 2017). Further, neophytes classified as invasive species homogenized floras while non-invasive neophytes differentiated floras, suggesting an effect of invasiveness as well as time since establishment (Ricotta et al. 2014). Place of origin may also be influential. For example, the similarity of Mexican and Spanish urban floras was increased mostly through bi-directional exchange between the two locations rather than through introduction of the same non-natives to both regions (Garcillán et al. 2014).

While most work on duration of establishment comes from European cities, research in the United States has found greater homogeneity among extralimital natives (species introduced from nearby sources, likely longer ago) than among non-natives from distant sources (La Sorte and McKinney 2006, La Sorte et al. 2007). Additionally, research in New Zealand showed greater community similarity among non-native plant species with longer residence times (Ricotta et al. 2017). Comparing floras in European and non-European cities globally showed no difference between native and non-native similarity, but greater homogeneity of European archaeophytes and invasive species in both European and non-European cities, suggesting that spread of archaeophytes and invasive species may lead to intercontinental homogenization (La Sorte et al. 2014a). Thus, there is robust support for the idea that invasive species and non-native species with longer durations of residence drive homogenization of urban floras.

Other studies compared the effects of native and non-native species without differentiating by time of introduction, with variable results. Non-native species were more similar than natives across 15 Chinese towns (Wang et al. 2014). In forest plots, native understory plant species homogenized more in response to surrounding urbanization than did non-native species (Trentanovi et al. 2013). Others found no evidence of urban homogenization by non-native species (Pino et al. 2009, Aronson et al. 2014, Yang et al. 2015). Given the divergent effects of recently introduced and established non-native species, limited overall effects of non-native species on community similarity are to be expected.

Another element of floristic studies not found in other urban homogenization studies is the distinction between cultivated plants (i.e., planted and managed) and spontaneous plants (i.e., dispersing and growing naturally). 59% of included vegetation studies considered only spontaneous plants, either by investigating remnant natural communities (e.g., Rogers et al. 2009, Trentanovi et al. 2013) or by excluding cultivated individuals from floristic surveys (e.g., Huwer and Wittig 2013, Lososová et al. 2016; Appendix A: Table S3). Thus, these studies examined only urban homogenization via establishment of non-native species as viable populations independent of human cultivation.

Studies specifically testing for homogenization of cultivated compared to spontaneous species reached mixed conclusions. In U.S. residential yards, both taxonomic and phylogenetic composition of cultivated and spontaneous species were more similar across regions than was the composition of natural vegetation (Pearse et al. 2018). Within the cultivated species pool, cultivated non-natives contributed to phylogenetic differentiation across regions while spontaneous non-natives contributed to phylogenetic homogenization (Padullés Cubino et al. 2019). However, vegetation in multiple urban habitats of 15 Chinese towns experienced homogenization by cultivated species and differentiation by spontaneous species (Wang et al. 2014). These results suggest different roles for cultivated and spontaneous species similar to the effects of recent and established invaders. Differing results in the U.S. and China suggest that a cultural or geographic component likely modifies these roles, potentially due to differing landscaping preferences, horticultural species pools, and native species suitability to the urban environment. Direct management of cultivated plant species complicates the urban homogenization hypothesis and must be further explored in order to more fully characterize similarity among urban environments.

Question 3: Homogenization relative to what?

Unlike research in biotic homogenization, which tests for increasing similarity over time (Olden et al. 2018), the most appropriate comparisons to test for urban homogenization are unclear. Most studies compared similarity along an urban to rural gradient, considering two or more different habitat types with varying levels of urbanity, disturbance, and/or management (Figure 3; Table 4). Some considered patches of natural or semi-natural vegetation and compared them based on metrics of surrounding urbanization (e.g., Johnson et al. 2013, Uchida et al. 2018). 50% of studies in natural patches found no evidence of urban homogenization, compared to 13% of land use gradient-type studies. Eight studies compared entire cities, usually to test for homogenization as increased similarity among non-natives compared to natives. Similarly, six studies compared entire watersheds or grid cells across an urbanized region to test whether areas with more urban land cover had more homogeneous biota.

Among studies using an urban to rural gradient approach, some considered several different urban land use types. For example, Horsák et al. (2013) found that non-native snails increased homogeneity of urban snail communities overall, but when split by land use type, early and mid-successional semi-natural sites were homogenized while boulevards and residential sites were differentiated by non-natives. Jim and Chen (2008) found lower variance among tree communities in riverside parks and street verges than in urban parks. These comparisons suggest that specific land uses and management strategies may lead to different results for community similarity.

For studies that took an urban to rural gradient approach, observing homogenization of urban areas may depend on characteristics of the "rural" end of the gradient. Several studies included agriculture as one of several comparison landscape types, with two studies comparing only against agriculture (Öckinger et al. 2009, Knop 2016). As agriculture can also result in biotic homogenization (Vellend et al. 2007, Ekroos et al. 2010, Rodrigues et al. 2013), comparisons against these landscapes may be misleading. However, several studies found that fauna in cities can be even more homogenized than in agricultural areas (butterflies - Öckinger et al. 2009, birds - Filloy et al. 2015, true bugs and leafhoppers - Knop 2016, bees - Harrison et al. 2018). This may not hold true for all taxa and locations, and may depend on the intensity of agricultural activity (Johnson et al. 2013).

While many comparisons were made among cities in the same biome, some studies explicitly tested whether urbanization led to greater similarity among different biomes. For example, bird communities in settlements in oak and deciduous forest were more similar than bird communities in the natural forest biomes (Vázquez-Reyes et al. 2017). Similarly, studies of the urban forest found that urban tree composition in different biomes was more dissimilar than was composition for cities in the same biome, which were fairly homogeneous (Nock et al. 2013, Ramage et al. 2013, Yang et al. 2015).

Question 4: Homogenization at what spatial scale?

Of the 57 included papers, 15 (26%) addressed urban homogenization in a single city, 28 (49%) in multiple cities or a larger region in one country, 6 (11%) in multiple countries on one continent, and 8 (14%) on multiple continents (Table 4). Most studies took place in North America (30, 53%) or Europe (21, 37%), with fewer including South America (10), Asia (8), Oceania (5), or Africa (4). Two studies included observations on all six urbanized continents (Aronson et al. 2014, Newbold et al. 2018).

Urban homogenization varied based on spatial scale, with some studies arguing that global convergence is in progress while others suggest that it is likely to be more limited. For example, studies of regional compared to global-scale homogenization of urban plant communities found greater support for convergence at regional scales, and suggested that global homogenization is unlikely (La Sorte et al. 2007, Yang et al. 2015). Yang et al. (2015) found that while most urban tree communities clustered regionally, one cluster contained cities from many regions, suggesting that shared culture or histories may cause some cities to converge while others remain regionally distinct. Of the two global-scale, multi-taxa studies, Newbold et al. (2018) supported homogenization by showing an increase in species with larger range sizes, while Aronson et al. (2014) did not, finding that plant and bird communities clustered more closely by geographic location than by urban or non-urban position. At a smaller spatial scale, Jenerette et al. (2006) showed reduced variability in soil characteristics within urban parks compared to desert patches but increased variability between urban parks. Overall, urban homogenization is best supported at the local to regional level, likely in response to shared management and climate.

26% of studies tested urban homogenization within a single city, rather than convergence across cities. These studies tested for homogenization among patches that share climatic and geographic characteristics and differ primarily in management and disturbance. Most compared beta diversity along a gradient of urbanization (see Question 3). This group also includes the four studies of temporal homogenization, which tested for decreased variability of communities over time rather than space (Leveau et al. 2015, Leveau and Leveau 2016, Kale et al. 2018, Uchida et al. 2018). Single-city studies showed mixed results, likely due to the variability of management and land use history within and between cities (Table 4). However, 73% of these studies found some evidence of homogenization, similar to 71% for studies of multiple cities in the same country.

Question 5: Homogenization by what mechanisms?

While biotic homogenization by the spread of non-natives and loss of natives was the primary mechanism of urban homogenization explored, some papers went further in investigating why urbanization led to increased similarity or how non-natives came to replace natives. Proposed drivers of urban homogenization included similar resource availability, management behaviors, disturbance regimes, human preferences, and horticultural sources. Each of these drivers could promote similar sets of species that are well adapted to the resulting environment, whether native or non-native. In some cases, these mechanisms can also explain convergence in the abiotic environment (e.g., shared preferences, management, and disturbance regimes). Similarities in urban abiotic conditions likely result in functional filtering, or the increased success of a subset of species with suitable functional traits for the environment along with suppression of functional groups that are less suitable (Duncan et al. 2011, Williams et al. 2015). Abiotic conditions and cultivated plant communities are most likely to experience homogenization through direct alteration and management, while other biota respond to these actions indirectly.

Management across urban land uses was often explored as a mechanism driving homogenization. For example, urban parks with more intensive management had more similar spontaneous non-woody plant communities, while park position along an urban-rural gradient did not predict community similarity (Fischer et al. 2016). Reduced weed management intensity was suggested as a driver of homogenization over time for spontaneous urban plant communities in German towns, suggesting a negative rather than positive relationship between management intensity and plant similarity (Huwer and Wittig 2013). Zooplankton communities in urban ponds with algae management treatments were less variable than those in untreated ponds, showing homogenization with management action (Sokol et al. 2015). Studies of soil biogeochemistry and microbial communities suggested strong influences of management behavior based on the spatial scales and subjects for which homogenization was supported (Jenerette et al. 2006, Pouyat et al. 2015, Epp Schmidt et al. 2017). Together, these studies support the hypothesis that direct management behavior is an important driver of urban homogenization. However, homogenization of taxa like birds which are not directly managed suggests that indirect mechanisms are also influential.

One indirect effect proposed as a driver of urban homogenization for birds was resource buffering, or supplemental provision of food and other resources throughout the year. Resource buffering was suggested as a mechanism for temporal homogenization, or increased similarity across seasons in urban areas compared to natural areas. This mechanism was well supported for birds (La Sorte et al. 2014b, Leveau et al. 2015, Leveau and Leveau 2016, Uchida et al. 2018), but needs further exploration for other taxa that may also be buffered from environmental variability by urban provisioning.

Few ecological studies tested for homogenization of human behavior, although other fields have dealt extensively with identifying behavioral similarities (e.g., marketing research, economics, geography, cultural studies, urban studies). While we excluded studies of human behavior from the final sample, some studies did investigate homogenization of residential yard management behavior across regions. Fertilization and irrigation behaviors were similar across cities, despite variability in precipitation and nutrient availability (Groffman et al. 2016). Behavior may differentiate more by socioeconomic status and life stage than by geographic position, thus resulting in within-city variability (Polsky et al. 2014). These behaviors partially explain the within-parcel and regional scales of homogenization observed across studies.

Plant preferences and horticultural availability could potentially promote or reduce homogenization of urban floras but were rarely investigated in this body of literature. In residential yards, preferences can lead to similar plant composition through selection of species with shared traits, resulting in phylogenetic homogenization (Kendal et al. 2012, Padullés Cubino et al. 2019). Differences in tree communities across parks may reflect park age, and thus the number of park managers and landscape fashions that have influenced a particular community (Jim and Chen 2008). Within the U.S., yard management priorities are shared across regions, suggesting similarity in the motivations underlying yard management behaviors (Larson et al. 2016). However, shared preferences may break down across social lifestyles, and thus, continued research is needed to explore cross-cultural homogenization of preferences and management behavior (Grove et al. 2014).

Related areas of research

Several themes arose from papers that contained relevant analyses and questions but did not meet all criteria for inclusion. While we cannot exhaustively review these themes, we highlight some here for future exploration.

Many urban-rural gradient studies included ordination analyses and other compositional comparisons, but did not test for changes in within-habitat variability or beta diversity. Evidence of differences between urban and non-urban spaces (e.g., change in species richness, differences in pollutant concentrations) do not necessarily address the question of increasing similarity, though they may support homogenization by showing shared characteristics of cities. Future research addressing more targeted questions (e.g., homogenization in residential yards, homogenization of urban birds) could include these findings to more fully evaluate all available evidence.

Genetic homogenization (i.e., increased relatedness among individuals in urban environments) is referenced in some included papers but not fully represented by this sample due to differences in terminology used in these studies. Two studies framed as testing urban genetic homogenization found greater relatedness among urban compared to suburban house sparrows (Vangestel et al. 2011) and among urban mosquitos compared to those in an outlying conservation area (Wilke et al. 2018). However, genetic studies more frequently tested the effects of fragmentation and reduced connectivity among urban populations. For example, Miles et al. (2018) found higher genetic diversity within urban spider populations but less diversity between populations, suggesting connectivity and gene flow. Research on genetic connectivity may reveal additional insight into population-level urban homogenization.

Research on behavioral plasticity and adaptation to the urban environment sometimes included urban homogenization framing. For example, bird song and personality boldness were less variable among urban individuals than among individuals in rural or natural habitats, showing evidence of behavioral homogenization (Møller 2010, Laiolo 2011, Potvin and Parris 2012). Beyond birds, cobras in urban environments had narrower, more homogeneous diets than cobras in suburban areas, potentially reflecting changes in homogeneity of urban prey populations as well as snake foraging behavior (Luiselli et al. 2002). Additional research on urban behavioral syndromes and adaptations suggests differences in behavior between urban and non-urban populations, but also decreased variability in urban population behavior.

Trait filtering in urban environments also suggests functional homogenization. Many studies have identified suites of traits that allow species to persist in cities (Knapp et al. 2008, Thompson and McCarthy 2008, Duncan et al. 2011, Sol et al. 2014). These studies typically don't measure or test homogenization, but the existence of shared functional traits that characterize urban communities suggests that functional homogenization is likely.

As research on urban homogenization expands to include abiotic elements of cities, overlap with other bodies of literature on characteristics of the urban environment is likely to increase. For example, convergence of the urban microclimate shares conceptual overlap with studies describing a common urban heat island effect across cities (e.g., Imhoff et al. 2010, Hall et al. 2016). Numerous studies have revealed altered hydrological and biogeochemical cycling in cities (e.g., Walsh et al. 2005, Kaye et al. 2006, Pickett et al. 2011), which could be thought of as convergence among cities but typically is explored by testing for common characteristics of cities rather than through testing for reduced variance among cities relative to natural spaces. The field of landscape ecology has explored urban landscape fragmentation and heterogeneity, showing that urban areas are more fragmented with smaller patch sizes, greater edge-to-area ratios, and higher land cover diversity (Wu et al. 2011, Li et al. 2013). These changes in the urban environment could be considered physical homogenization, and may also explain patterns of biotic homogenization (Aronson et al. 2014). As studies of urban homogenization expand, care is needed to ensure appropriate use of existing literature in diverse areas with varying terminology.

CONCLUSIONS

Overall, we found a significant body of evidence supporting the urban homogenization hypothesis. As expected, there was variation in this pattern by spatial scale, subject or taxa, and type of comparison. Defining homogenization as decreased variability, increased similarity, increased non-native species abundance, or increased cosmopolitan species resulted in different study approaches, but often yielded similar conclusions. Phylogenetic, functional, and temporal homogenization were supported by the few studies testing these types of convergence. Urban homogenization was clearly supported for birds but not for beetles, and plant communities were homogenized by established non-native species. Other taxa remain understudied, including mammals, reptiles, amphibians, microbes, and aquatic organisms.

As we hypothesized, homogenization was more often supported in directly modified urban areas than in natural patches and was more likely in landscapes with common management patterns and at spatial scales with shared biophysical conditions. Unmanaged taxa like birds that benefit from direct resource provisioning and other choices also showed homogenization as an indirect result of human actions. Management behaviors are an important area for future exploration of the mechanisms driving urban homogenization. However, the spread of non-native and invasive species in urban areas is also an important driver to consider, and human management may be able to preserve local uniqueness while unmanaged communities tend towards similarity over time (Olden et al. 2018). Researchers disagree about the potential for global-scale urban homogenization, which may be prevented by climatic variability and variation in management across cultural, socioeconomic, historical, and political contexts. Thus, the impact of urban homogenization on biodiversity at a global scale may be more limited than feared.

Future research should continue to explore potential mechanisms for urban homogenization, while drawing on the diverse and growing existing literature. Further investigation into understudied taxa and locations will improve our understanding of this hypothesis, as will comparisons across management regimes. As urban areas continue to grow globally, continuing evaluation of the threats posed to global biodiversity by anthropogenic activity remains essential to the preservation of healthy ecosystems.

Additionally, place-based ecologies in cities have the potential to deliver cultural benefits to urban residents, but require the maintenance of locally unique biodiversity.

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REFERENCES

Adler, F. R., and C. J. Tanner. 2013. Urban ecosystems: ecological principles for the built environment. Cambridge University Press, New York, New York, USA.

Anderson, M. J., T. O. Crist, J. M. Chase, M. Vellend, B. D. Inouye, A. L. Freestone, N. J. Sanders, H. V. Cornell, L. S. Comita, K. F. Davies, S. P. Harrison, N. J. B. Kraft, J. C. Stegen, and N. G. Swenson. 2011. Navigating the multiple meanings of β diversity: a roadmap for the practicing ecologist. Ecology Letters 14:19–28.

Aronson, M. F. J., S. N. Handel, I. P. La Puma, and S. E. Clemants. 2015. Urbanization promotes non-native woody species and diverse plant assemblages in the New York metropolitan region. Urban Ecosystems 18:31–45.

Aronson, M. F. J., F. A. La Sorte, C. H. Nilon, M. Katti, M. A. Goddard, C. A. Lepczyk, P. S. Warren, N. S. G. Williams, S. Cilliers, B. Clarkson, C. Dobbs, R. Dolan, M. Hedblom, S. Klotz, J. L. Kooijmans, I. Kühn, I. MacGregor-Fors, M. McDonnell, U. Mörtberg, P. Pyšek, S. Siebert, J. Sushinsky, P. Werner, and M. Winter. 2014. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. Proceedings of the Royal Society B: Biological Sciences 281:20133330.

Asmus, U., and G. L. Rapson. 2014. Floristic homogeneity underlies environmental diversification of northern New Zealand urban areas. New Zealand Journal of Botany 52:285–303.

Baselga, A. 2010. Partitioning the turnover and nestedness components of beta diversity. Global Ecology and Biogeography 19:134–143.

Bertocci, I., F. Arenas, E. Cacabelos, G. M. Martins, M. I. Seabra, N. V. Álvaro, J. N. Fernandes, R. Gaião, N. Mamede, M. Mulas, and A. I. Neto. 2017. Nowhere safe? Exploring the influence of urbanization across mainland and insular seashores in continental Portugal and the Azorean Archipelago. Marine Pollution Bulletin 114:644–655.

Blair, R. 2004. The effects of urban sprawl on birds at multiple levels of biological organization. Ecology and Society 9:2.

Blair, R. B., and E. M. Johnson. 2008. Suburban habitats and their role for birds in the urban-rural habitat network: points of local invasion and extinction? Landscape Ecology 23:1157–1169.

Bradley, B. A., D. M. Blumenthal, R. Early, E. D. Grosholz, J. J. Lawler, L. P. Miller, C. J. B. Sorte, C. M. D’Antonio, J. M. Diez, J. S. Dukes, I. Ibanez, and J. D. Olden. 2012. Global change, global trade, and the next wave of plant invasions. Frontiers in Ecology and the Environment 10:20–28.

Brice, M.-H., S. Pellerin, and M. Poulin. 2017. Does urbanization lead to taxonomic and functional homogenization in riparian forests? Diversity and Distributions 23:828–840.

Cardinale, B. J., J. E. Duffy, A. Gonzalez, D. U. Hooper, C. Perrings, P. Venail, A. Narwani, G. M. Mace, D. Tilman, D. A. Wardle, A. P. Kinzig, G. C. Daily, M. Loreau, J. B. Grace, A. Larigauderie, D. S. Srivastava, and S. Naeem. 2012. Biodiversity loss and its impact on humanity. Nature 486:59–67.

Catterall, C. P., J. A. Cousin, S. Piper, and G. Johnson. 2010. Long-term dynamics of bird diversity in forest and suburb: decay, turnover or homogenization? Diversity and Distributions 16:559–570.

Deák, B., B. Hüse, and B. Tóthmérész. 2016. Grassland vegetation in urban habitats - testing ecological theories. Tuexenia 36:379–393.

Docherty, K. M., D. S. Pearce, K. M. Lemmer, and R. L. Hale. 2018. Distributing regionally, distinguishing locally: examining the underlying effects of local land use on airborne bacterial biodiversity. Environmental Microbiology 20:3529–3542.

Duncan, R. P., S. E. Clemants, R. T. Corlett, A. K. Hahs, M. A. McCarthy, M. J. McDonnell, M. W. Schwartz, K. Thompson, P. A. Vesk, and N. S. G. Williams. 2011. Plant traits and extinction in urban areas: a meta-analysis of 11 cities. Global Ecology and Biogeography 20:509–519.

Ekroos, J., J. Heliölä, and M. Kuussaari. 2010. Homogenization of lepidopteran communities in intensively cultivated agricultural landscapes. Journal of Applied Ecology 47:459–467.

Epp Schmidt, D. J., R. Pouyat, K. Szlavecz, H. Setälä, D. J. Kotze, I. Yesilonis, S. Cilliers, E. Hornung, M. Dombos, and S. A. Yarwood. 2017. Urbanization erodes ectomycorrhizal fungal diversity and may cause microbial communities to converge. Nature Ecology & Evolution 1:0123.

Filloy, J., S. Grosso, and M. I. Bellocq. 2015. Urbanization altered latitudinal patterns of bird diversity-environment relationships in the southern Neotropics. Urban Ecosystems 18:777–791.

Fischer, L. K., V. Rodorff, M. von der Lippe, and I. Kowarik. 2016. Drivers of biodiversity patterns in parks of a growing South American megacity. Urban Ecosystems 19:1231–1249.

Garcillán, P. P., E. D. Dana, J. P. Rebman, and J. Peñas. 2014. Effects of alien species on homogenization of urban floras across continents: a tale of two mediterranean cities in two different continents. Plant Ecology and Evolution 147:3–9.

Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai, and J. M. Briggs. 2008. Global change and the ecology of cities. Science 319:756–760.

Groffman, P. M., M. Avolio, J. Cavender-Bares, N. D. Bettez, J. M. Grove, S. J. Hall, S. E. Hobbie, K. L. Larson, S. B. Lerman, D. H. Locke, J. B. Heffernan, J. L. Morse, C. Neill, K. C. Nelson, J. O'Neil-Dunne, D. E. Pataki, C. Polsky, R. Roy Chowdhury, and T. L. E. Trammell. 2017. Ecological homogenization of residential macrosystems. Nature Ecology & Evolution 1:0191.

Groffman, P. M., J. Cavender-Bares, N. D. Bettez, J. M. Grove, S. J. Hall, J. B. Heffernan, S. E. Hobbie, K. L. Larson, J. L. Morse, C. Neill, K. Nelson, J. O'Neil-Dunne, L. Ogden, D. E. Pataki, C. Polsky, R. Roy Chowdhury, and M. K. Steele. 2014. Ecological homogenization of urban USA. Frontiers in Ecology and the Environment 12:74–81.

Groffman, P. M., J. M. Grove, C. Polsky, N. D. Bettez, J. L. Morse, J. Cavender-Bares, S. J. Hall, J. B. Heffernan, S. E. Hobbie, K. L. Larson, C. Neill, K. Nelson, L. Ogden, J. O'Neil-Dunne, D. Pataki, R. Roy Chowdhury, and D. H. Locke. 2016. Satisfaction, water and fertilizer use in the American residential macrosystem. Environmental Research Letters 11:034004.

Grove, J. M., D. H. Locke, and J. P. M. O'Neil-Dunne. 2014. An ecology of prestige in New York City: examining the relationships among population density, socio-economic status, group identity, and residential canopy cover. Environmental Management 54:402–419.

Hall, S. J., J. Learned, B. Ruddell, K. L. Larson, J. Cavender-Bares, N. Bettez, P. M. Groffman, J. M. Grove, J. B. Heffernan, S. E. Hobbie, J. L. Morse, C. Neill, K. C. Nelson, J. P. M. O'Neil-Dunne, L. Ogden, D. E. Pataki, W. D. Pearse, C. Polsky, R. Roy Chowdhury, M. K. Steele, and T. L. E. Trammell. 2016. Convergence of microclimate in residential landscapes across diverse cities in the United States. Landscape Ecology 31:101–117.

Harrison, T., J. Gibbs, and R. Winfree. 2018. Phylogenetic homogenization of bee communities across ecoregions. Global Ecology and Biogeography 27:1457–1466.

Hartley, D. J., M. J. Koivula, J. R. Spence, R. Pelletier, and G. E. Ball. 2007. Effects of urbanization on ground beetle assemblages (Coleoptera, Carabidae) of grassland habitats in western Canada. Ecography 30:673–684.

Hausmann, A., R. Slotow, J. K. Burns, and E. Di Minin. 2016. The ecosystem service of sense of place: benefits for human well-being and biodiversity conservation. Environmental Conservation 43:117–127.

Holway, D. A., and A. V. Suarez. 2006. Homogenization of ant communities in mediterranean California: the effects of urbanization and invasion. Biological Conservation 127:319–326.

Horsák, M., T. Čejka, L. Juřičková, V. Wiese, V. Horsáková, and Z. Lososová. 2016. Drivers of Central European urban land snail faunas: the role of climate and local species pool in the representation of native and non-native species. Biological Invasions 18:3547–3560.

Horsák, M., Z. Lososová, T. Čejka, L. Juřičková, and M. Chytrý. 2013. Diversity and biotic homogenization of urban land-snail faunas in relation to habitat types and macroclimate in 32 Central European cities. PLoS ONE 8:e71783.

Huwer, A., and R. Wittig. 2013. Evidence for increasing homogenization and de-ruralization of the Central European village flora. Tuexenia 33:213–231.

Imhoff, M. L., P. Zhang, R. E. Wolfe, and L. Bounoua. 2010. Remote sensing of the urban heat island effect across biomes in the continental USA. Remote Sensing of Environment 114:504–513.

Jenerette, G. D., J. Wu, N. B. Grimm, and D. Hope. 2006. Points, patches, and regions: scaling soil biogeochemical patterns in an urbanized arid ecosystem. Global Change Biology 12:1532–1544.

Jim, C. Y., and W. Y. Chen. 2008. Pattern and divergence of tree communities in Taipei's main urban green spaces. Landscape and Urban Planning 84:312–323.

Johnson, P. T. J., J. T. Hoverman, V. J. McKenzie, A. R. Blaustein, and K. L. D. Richgels. 2013. Urbanization and wetland communities: applying metacommunity theory to understand the local and landscape effects. Journal of Applied Ecology 50:34–42.

Kale, M., N. Dudhe, M. Ferrante, T. Ivanova, R. Kasambe, I. S. Trukhanova, P. Bhattacharya, and G. L. Lövei. 2018. The effect of urbanization on the functional and scale-sensitive diversity of bird assemblages in Central India. Journal of Tropical Ecology 34:341–350.

Kaye, J. P., P. M. Groffman, N. B. Grimm, L. A. Baker, and R. V. Pouyat. 2006. A distinct urban biogeochemistry? Trends in Ecology & Evolution 21:192–199.

Kendal, D., K. J. H. Williams, and N. S. G. Williams. 2012. Plant traits link people's plant preferences to the composition of their gardens. Landscape and Urban Planning 105:34–42.

Knapp, S., I. Kühn, R. Wittig, W. A. Ozinga, P. Poschlod, and S. Klotz. 2008. Urbanization causes shifts in species' trait state frequencies. Preslia 80:375–388.

Knapp, S., and R. Wittig. 2012. An analysis of temporal homogenisation and differentiation in Central European village floras. Basic and Applied Ecology 13:319–327.

Knop, E. 2016. Biotic homogenization of three insect groups due to urbanization. Global Change Biology 22:228–236.

Koricheva, J., J. Gurevitch, and K. Mengersen, editors. 2013. Handbook of meta-analysis in ecology and evolution. Princeton University Press.

La Sorte, F. A., M. F. J. Aronson, N. S. G. Williams, L. Celesti-Grapow, S. Cilliers, B. D. Clarkson, R. W. Dolan, A. Hipp, S. Klotz, I. Kühn, P. Pyšek, S. Siebert, and M. Winter. 2014a. Beta diversity of urban floras among European and non-European cities. Global Ecology and Biogeography 23:769–779.

La Sorte, F. A., and M. L. McKinney. 2006. Compositional similarity and the distribution of geographical range size for assemblages of native and non-native species in urban floras. Diversity and Distributions 12:679–686.

La Sorte, F. A., M. L. McKinney, and P. Pyšek. 2007. Compositional similarity among urban floras within and across continents: biogeographical consequences of human-mediated biotic interchange. Global Change Biology 13:913–921.

La Sorte, F. A., M. L. McKinney, P. Pyšek, S. Klotz, G. L. Rapson, L. Celesti-Grapow, and K. Thompson. 2008. Distance decay of similarity among European urban floras: the impact of anthropogenic activities on β diversity. Global Ecology and Biogeography 17:363–371.

La Sorte, F. A., M. W. Tingley, and A. H. Hurlbert. 2014b. The role of urban and agricultural areas during avian migration: an assessment of within-year temporal turnover. Global Ecology and Biogeography 23:1225–1234.

Laiolo, P. 2011. Homogenisation of birdsong along a natural-urban gradient in Argentina. Ethology Ecology & Evolution 23:274–287.

Larson, K. L., K. C. Nelson, S. R. Samples, S. J. Hall, N. Bettez, J. Cavender-Bares, P. M. Groffman, M. Grove, J. B. Heffernan, S. E. Hobbie, J. Learned, J. L. Morse, C. Neill, L. A. Ogden, J. O'Neil-Dunne, D. E. Pataki, C. Polsky, R. Roy Chowdhury, M. Steele, and T. L. E. Trammell. 2016. Ecosystem services in managing residential landscapes: priorities, value dimensions, and cross-regional patterns. Urban Ecosystems 19:95–113.

Leveau, L. M., F. I. Isla, and M. I. Bellocq. 2015. Urbanization and the temporal homogenization of bird communities: a case study in central Argentina. Urban Ecosystems 18:1461–1476.

Leveau, L. M., and C. M. Leveau. 2016. Does urbanization affect the seasonal dynamics of bird communities in urban parks? Urban Ecosystems 19:631–647.

Li, C., J. Li, and J. Wu. 2013. Quantifying the speed, growth modes, and landscape pattern changes of urbanization: a hierarchical patch dynamics approach. Landscape Ecology 28:1875–1888.

Lososová, Z., M. Chytrý, J. Danihelka, L. Tichý, and C. Ricotta. 2016. Biotic homogenization of urban floras by alien species: the role of species turnover and richness differences. Journal of Vegetation Science 27:452–459.

Lososová, Z., M. Chytrý, L. Tichý, J. Danihelka, K. Fajmon, O. Hájek, K. Kintrová, D. Láníková, Z. Otýpková, and V. Řehořek. 2012. Biotic homogenization of Central

European urban floras depends on residence time of alien species and habitat types. Biological Conservation 145:179–184.

Luck, G. W., and L. T. Smallbone. 2011. The impact of urbanization on taxonomic and functional similarity among bird communities. Journal of Biogeography 38:894–906.

Luiselli, L., F. M. Angelici, and G. C. Akani. 2002. Comparative feeding strategies and dietary plasticity of the sympatric cobras *Naja melanoleuca* and *Naja nigricollis* in three diverging Afrotropical habitats. Canadian Journal of Zoology 80:55–63.

Magura, T., G. L. Lövei, and B. Tóthmérész. 2010. Does urbanization decrease diversity in ground beetle (Carabidae) assemblages? Global Ecology and Biogeography 19:16–26.

Marchetti, M. P., J. L. Lockwood, and T. Light. 2006. Effects of urbanization on California's fish diversity: differentiation, homogenization and the influence of spatial scale. Biological Conservation 127:310–318.

McKinney, M. L. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127:247–260.

McKinney, M. L., and J. L. Lockwood. 1999. Biotic homogenization: a few winners replacing many losers in the next mass extinction. Trends in Ecology & Evolution 14:450–453.

Miles, L. S., J. C. Johnson, R. J. Dyer, and B. C. Verrelli. 2018. Urbanization as a facilitator of gene flow in a human health pest. Molecular Ecology 27:3219–3230.

Møller, A. P. 2010. Interspecific variation in fear responses predicts urbanization in birds. Behavioral Ecology 21:365–371.

Morelli, F., Y. Benedetti, J. D. Ibáñez-Álamo, J. Jokimäki, R. Mänd, P. Tryjanowski, and A. P. Møller. 2016. Evidence of evolutionary homogenization of bird communities in urban environments across Europe. Global Ecology and Biogeography 25:1284–1293.

Murthy, A. C., T. S. Fristoe, and J. R. Burger. 2016. Homogenizing effects of cities on North American winter bird diversity. Ecosphere 7:e01216.

Newbold, T., L. N. Hudson, S. Contu, S. L. L. Hill, J. Beck, Y. Liu, C. Meyer, H. R. P. Phillips, J. P. W. Scharlemann, and A. Purvis. 2018. Widespread winners and narrow-ranged losers: land use homogenizes biodiversity in local assemblages worldwide. PLoS Biology 16:e2006841.

Nock, C. A., A. Paquette, M. Follett, D. J. Nowak, and C. Messier. 2013. Effects of urbanization on tree species functional diversity in eastern North America. Ecosystems 16:1487–1497.

Öckinger, E., Å. Dannestam, and H. G. Smith. 2009. The importance of fragmentation and habitat quality of urban grasslands for butterfly diversity. Landscape and Urban Planning 93:31–37.

Olden, J. D. 2006. Biotic homogenization: a new research agenda for conservation biogeography. Journal of Biogeography 33:2027–2039.

Olden, J. D., L. Comte, and X. Giam. 2018. The Homogocene: a research prospectus for the study of biotic homogenisation. NeoBiota 37:23–36.

Oliveira, J. P., I. Sousa-Pinto, G. M. Weber, and I. Bertocci. 2014. Urban vs. extra-urban environments: scales of variation of intertidal benthic assemblages in north Portugal. Marine Environmental Research 97:48–57.

Padullés Cubino, J., J. Cavender-Bares, S. E. Hobbie, S. J. Hall, T. L. E. Trammell, C. Neill, M. L. Avolio, L. E. Darling, and P. M. Groffman. 2019. Contribution of non-native plants to the phylogenetic homogenization of U.S. yard floras. Ecosphere 10:e02638.

Patitucci, L. D., P. R. Mulieri, J. A. Schnack, and J. C. Mariluis. 2011. Species composition and heterogeneity of blowflies assemblages (Diptera: Calliphoridae) in urban-rural gradients at regional scale in Argentinean Patagonia. Studies on Neotropical Fauna and Environment 46:49–58.

Pearse, W. D., J. Cavender-Bares, S. E. Hobbie, M. L. Avolio, N. Bettez, R. Roy Chowdhury, L. E. Darling, P. M. Groffman, J. M. Grove, S. J. Hall, J. B. Heffernan, J. Learned, C. Neill, K. C. Nelson, D. E. Pataki, B. L. Ruddell, M. E. Steele, and T. L. E. Trammell. 2018. Ecological homogenisation in North American urban yards: vegetation diversity, composition, and structure. Ecosphere 9:e02105.

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

Pino, J., X. Font, M. de Cáceres, and R. Molowny-Horas. 2009. Floristic homogenization by native ruderal and alien plants in north-east Spain: the effect of environmental differences on a regional scale. Global Ecology and Biogeography 18:563–574.

Polsky, C., J. M. Grove, C. Knudson, P. M. Groffman, N. Bettez, J. Cavender-Bares, S. J. Hall, J. B. Heffernan, S. E. Hobbie, K. L. Larson, J. L. Morse, C. Neill, K. C. Nelson,

L. A. Ogden, J. O'Neil-Dunne, D. E. Pataki, R. Roy Chowdhury, and M. K. Steele. 2014. Assessing the homogenization of urban land management with an application to US residential lawn care. Proceedings of the National Academy of Sciences USA 111:4432–4437.

Pool, T. K., and J. D. Olden. 2012. Taxonomic and functional homogenization of an endemic desert fish fauna. Diversity and Distributions 18:366–376.

Potvin, D. A., and K. M. Parris. 2012. Song convergence in multiple urban populations of silvereyes (*Zosterops lateralis*). Ecology and Evolution 2:1977–1984.

Pouyat, R. V., I. D. Yesilonis, M. Dombos, K. Szlavecz, H. Setälä, S. Cilliers, E. Hornung, D. J. Kotze, and S. Yarwood. 2015. A global comparison of surface soil characteristics across five cities: a test of the urban ecosystem convergence hypothesis. Soil Science 180:136–145.

Rahel, F. J. 2000. Homogenization of fish faunas across the United States. Science 288:854–856.

Ramage, B. S., L. A. Roman, and J. S. Dukes. 2013. Relationships between urban tree communities and the biomes in which they reside. Applied Vegetation Science 16:8–20.

Reichert, B. E., A. R. Sovie, B. J. Udell, K. M. Hart, R. R. Borkhataria, M. Bonneau, R. Reed, and R. McCleery. 2017. Urbanization may limit impacts of an invasive predator on native mammal diversity. Diversity and Distributions 23:355–367.

Ricotta, C., L. Celesti-Grapow, I. Kühn, G. Rapson, P. Pyšek, F. A. La Sorte, and K. Thompson. 2014. Geographical constraints are stronger than invasion patterns for European urban floras. PLoS ONE 9:e85661.

Ricotta, C., G. L. Rapson, U. Asmus, P. Pyšek, I. Kühn, F. A. La Sorte, and K. Thompson. 2017. British plants as aliens in New Zealand cities: residence time moderates their impact on the beta diversity of urban floras. Biological Invasions 19:3589–3599.

Rocha, A. M., A. González-Reyes, J. Corronca, S. Rodríguez-Artigas, I. Doma, E. Y. Repp, and X. Acosta. 2016. Tardigrade diversity: an evaluation of natural and disturbed environments of the province of Salta (Argentina). Zoological Journal of the Linnean Society 178:755–764.

Rodrigues, J. L. M., V. H. Pellizari, R. Mueller, K. Baek, E. da C. Jesus, F. S. Paula, B. Mirza, G. S. Hamaou, S. M. Tsai, B. Feiglf, J. M. Tiedje, B. J. M. Bohannan, and K. Nüsslein. 2013. Conversion of the Amazon rainforest to agriculture results in biotic homogenization of soil bacterial communities. Proceedings of the National Academy of Sciences USA 110:988–993.

Rogers, D. A., T. P. Rooney, T. J. Hawbaker, V. C. Radeloff, and D. M. Waller. 2009. Paying the extinction debt in southern Wisconsin forest understories. Conservation Biology 23:1497–1506.

Rooney, T. P., J. D. Olden, M. K. Leach, and D. A. Rogers. 2007. Biotic homogenization and conservation prioritization. Biological Conservation 134:447–450.

Scott, M. C. 2006. Winners and losers among stream fishes in relation to land use legacies and urban development in the southeastern US. Biological Conservation 127:301–309.

Sokol, E. R., B. L. Brown, C. C. Carey, B. M. Tornwall, C. M. Swan, and J. E. Barrett. 2015. Linking management to biodiversity in built ponds using metacommunity simulations. Ecological Modelling 296:36–45.

Sol, D., C. González-Lagos, D. Moreira, J. Maspons, and O. Lapiedra. 2014. Urbanisation tolerance and the loss of avian diversity. Ecology Letters 17:942–950.

Steele, M. K., J. B. Heffernan, N. Bettez, J. Cavender-Bares, P. M. Groffman, J. M. Grove, S. Hall, S. E. Hobbie, K. Larson, J. L. Morse, C. Neill, K. C. Nelson, J. O'Neil-Dunne, L. Ogden, D. E. Pataki, C. Polsky, and R. Roy Chowdhury. 2014. Convergent surface water distributions in U.S. cities. Ecosystems 17:685–697.

Thompson, K., and M. A. McCarthy. 2008. Traits of British alien and native urban plants. Journal of Ecology 96:853–859.

Trentanovi, G., M. von der Lippe, T. Sitzia, U. Ziechmann, I. Kowarik, and A. Cierjacks. 2013. Biotic homogenization at the community scale: disentangling the roles of urbanization and plant invasion. Diversity and Distributions 19:738–748.

Uchida, K., H. Fujimoto, and A. Ushimaru. 2018. Urbanization promotes the loss of seasonal dynamics in the semi-natural grasslands of an East Asian megacity. Basic and Applied Ecology 29:1–11.

Ulrich, W., M. Zalewski, and K. Komosiński. 2007. Diversity of carrion visiting beetles at rural and urban sites. Community Ecology 8:171–181.

Vangestel, C., J. Mergeay, D. A. Dawson, V. Vandomme, and L. Lens. 2011. Spatial heterogeneity in genetic relatedness among house sparrows along an urban–rural gradient as revealed by individual-based analysis. Molecular Ecology 20:4643–4653.

Vázquez-Reyes, L. D., M. del C. Arizmendi, H. O. Godínez-Álvarez, and A. G. Navarro-Sigüenza. 2017. Directional effects of biotic homogenization of bird communities in Mexican seasonal forests. The Condor 119:275–288.

Vellend, M., K. Verheyen, K. M. Flinn, H. Jacquemyn, A. Kolb, H. Van Calster, G. Peterken, B. J. Graae, J. Bellemare, O. Honnay, J. Brunet, M. Wulf, F. Gerhardt, and M. Hermy. 2007. Homogenization of forest plant communities and weakening of species-environment relationships via agricultural land use. Journal of Ecology 95:565–573.

Villegas Vallejos, M. A., A. A. Padial, and J. R. S. Vitule. 2016. Human-induced landscape changes homogenize Atlantic Forest bird assemblages through nested species loss. PLoS ONE 11:e0147058.

von der Lippe, M., and I. Kowarik. 2008. Do cities export biodiversity? Traffic as dispersal vector across urban-rural gradients. Diversity and Distributions 14:18–25.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgon. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706–723.

Wang, G., J. Zuo, X.-R. Li, Y. Liu, J. Yu, H. Shao, and Y. Li. 2014. Low plant diversity and floristic homogenization in fast-urbanizing towns in Shandong Peninsular, China: effects of urban greening at regional scale for ecological engineering. Ecological Engineering 64:179–185.

Wilke, A. B. B., G. C. de Carvalho, and M. T. Marrelli. 2018. Retention of ancestral polymorphism in *Culex nigripalpus* (Diptera: Culicidae) from São Paulo, Brazil. Infection, Genetics and Evolution 65:333–339.

Williams, N. S. G., A. K. Hahs, and P. A. Vesk. 2015. Urbanisation, plant traits and the composition of urban floras. Perspectives in Plant Ecology, Evolution and Systematics 17:78–86.

Wu, J., G. D. Jenerette, A. Buyantuyev, and C. L. Redman. 2011. Quantifying spatiotemporal patterns of urbanization: The case of the two fastest growing metropolitan regions in the United States. Ecological Complexity 8:1–8.

Yang, J., F. A. La Sorte, P. Pyšek, P. Yan, D. Nowak, and J. McBride. 2015. The compositional similarity of urban forests among the world’s cities is scale dependent. Global Ecology and Biogeography 24:1413–1423.

TABLE 1. Categories used to classify the research approach of each urban homogenization (UH) paper.

Types of UH	Measures of UH	Subjects	Comparisons	Spatial scales
▪ Taxonomic ▪ Phylogenetic ▪ Functional ▪ Structural ▪ Historical - taxonomic ▪ Historical - functional ▪ Temporal - taxonomic ▪ Abiotic	▪ Beta diversity ▪ Decomposed beta diversity ▪ Cluster analysis ▪ Distance to group average ▪ Similarity decay over geographic distance ▪ Relative similarity or abundance of native and non-native species ▪ Range size ▪ Variance	▪ Amphibians ▪ Aquatic invertebrates ▪ Birds ▪ Fish ▪ Mammals ▪ Microbes ▪ Plants ▪ Reptiles ▪ Rocky intertidal communities ▪ Terrestrial invertebrates	▪ Distance to urban area ▪ Entire cities ▪ Grid cells or watersheds ▪ Natural patches ▪ Land use gradient ▪ Single urban land use	▪ One city ▪ Multiple cities in one country ▪ Multiple countries on one continent ▪ Multiple continents

TABLE 2. Findings for different types of urban homogenization (UH). Studies testing multiple types of UH and are listed in all appropriate locations. Values in the evidence columns show the number and percentage of studies of each type of homogenization with each result. Total N = 57.

Type of UH	Evidence for UH	Mixed evidence	No evidence / evidence against UH
Taxonomic 42 studies (74%)	18 (43%) (3, 5, 11, 13, 17, 30, 33, 34, 35, 36, 37, 41, 45, 47, 49, 54, 55, 56)	11 (26%) (8, 9, 10, 18, 21, 24, 25, 29, 46, 51, 57)	13 (31%) (1, 2, 4, 6, 12, 15, 16, 22, 31, 38, 40, 42, 53)
Phylogenetic 4 studies (7%)	3 (75%) (15, 33, 41)	1 (25%) (39)	-
Functional 3 studies (5%)	2 (66%) (17, 30)	-	1 (33%) (6)
Structural 1 study (2%)	1 (100%) (41)	-	-
Historical - taxonomic 5 studies (9%)	3 (60%) (7, 19, 48)	-	2 (40%) (32, 43)
Historical - functional 1 study (2%)	-	-	1 (100%) (43)
Temporal - taxonomic 5 studies (9%)	5 (100%) (23, 26, 27, 28, 52)	-	-
Abiotic 4 studies (7%)	2 (50%) (14, 50)	2 (50%) (20, 44)	-

Notes: Small numerals indicate references. (1) Aronson et al. 2014; (2) Aronson et al. 2015; (3) Asmus and Rapson 2014; (4) Bertocci et al. 2017; (5) Blair and Johnson 2008; (6) Brice et al. 2017; (7) Catterall et al. 2010; (8) Deák et al. 2016; (9) Docherty et al. 2018; (10) Epp Schmidt et al. 2017; (11) Filloy et al. 2015; (12) Fischer et al. 2016; (13) Garcillán et al. 2014; (14) Hall et al. 2016; (15) Harrison et al. 2018; (16) Hartley et al. 2007; (17) Holway and Suarez 2006; (18) Horsák et al. 2013; (19) Huwer and Wittig 2013; (20) Jenerette et al. 2006; (21) Jim and Chen 2008; (22) Johnson et al. 2013; (23) Kale et al. 2018; (24) Knop 2016; (25) La Sorte et al. 2007; (26) La Sorte et al. 2014b; (27) Leveau and Leveau 2016; (28) Leveau et al 2015; (29) Lososová et al. 2012; (30) Luck and Smallbone 2011; (31) Magura et al. 2010; (32) Marchetti et al. 2006; (33) Morelli et al. 2016; (34) Murthy et al. 2016; (35) Newbold et al. 2018; (36) Nock et al. 2013; (37) Öckinger et al. 2009; (38) Oliveira et al. 2014; (39) Padullés Cubino et al. 2019; (40) Patitucci et al. 2011; (41) Pearse et al. 2018; (42) Pino et al. 2009; (43) Pool and Olden 2012; (44) Pouyat et al. 2015; (45) Reichert et al. 2017; (46) Ricotta et al. 2014; (47) Rocha et al. 2016; (48) Rogers et al. 2009; (49) Scott 2006; (50) Steele et al. 2014; (51) Trentanovi et al. 2013; (52) Uchida et al. 2018; (53) Ulrich et al. 2007; (54) Villegas Vallejos et al. 2016; (55) Vázquez-Reyes et al. 2017; (56) Wang et al. 2014; (57) Yang et al. 2015

TABLE 3. Types of measurement or analysis used to test for urban homogenization (UH). Some studies used multiple measurements and are listed more than once. Total N = 57.

Measurement type	Evidence for UH	Mixed evidence	No evidence / evidence against UH
Beta diversity 21 studies (37%)	10 (48%) (5, 7, 11, 17, 27, 28, 33, 37, 41, 48)	7 (33%) (9, 10, 15, 21, 51, 57)	4 (19%) (2, 12, 31, 32)
Decomposed beta diversity 16 studies (28%)	9 (56%) (3, 26, 30, 45, 47, 52, 54, 55, 56)	3 (19%) (19, 24, 25)	4 (25%) (6, 42, 43, 53)
Cluster analysis 8 studies (14%)	5 (63%) (5, 7, 23, 36, 55)	1 (13%) (57)	2 (25%) (1, 31)
Distance decay 9 studies (16%)	4 (44%) (3, 11, 30, 34)	3 (33%) (15, 24, 29)	2 (22%) (22, 40)
Distance to group average 4 studies (7%)	1 (25%) (54)	1 (25%) (46)	2 (50%) (6, 16)
Native/non-native difference 7 studies (12%)	2 (29%) (13, 49)	5 (71%) (8, 18, 29, 29, 46)	-
Range size 1 study (2%)	1 (100%) (35)	-	-
Variance 8 studies (14%)	4 (50%) (14, 28, 41, 50)	2 (25%) (20, 44)	2 (25%) (4, 38)

Notes: (1) Aronson et al. 2014; (2) Aronson et al. 2015; (3) Asmus and Rapson 2014; (4) Bertocci et al. 2017; (5) Blair and Johnson 2008; (6) Brice et al. 2017; (7) Catterall et al. 2010; (8) Deák et al. 2016; (9) Docherty et al. 2018; (10) Epp Schmidt et al. 2017; (11) Filloy et al. 2015; (12) Fischer et al. 2016; (13) Garcillán et al. 2014; (14) Hall et al. 2016; (15) Harrison et al. 2018; (16) Hartley et al. 2007; (17) Holway and Suarez 2006; (18) Horsák et al. 2013; (19) Huwer and Wittig 2013; (20) Jenerette et al. 2006; (21) Jim and Chen 2008; (22) Johnson et al. 2013; (23) Kale et al. 2018; (24) Knop 2016; (25) La Sorte et al. 2007; (26) La Sorte et al. 2014b; (27) Leveau and Leveau 2016; (28) Leveau et al 2015; (29) Lososová et al. 2012; (30) Luck and Smallbone 2011; (31) Magura et al. 2010; (32) Marchetti et al. 2006; (33) Morelli et al. 2016; (34) Murthy et al. 2016; (35) Newbold et al. 2018; (36) Nock et al. 2013; (37) Öckinger et al. 2009; (38) Oliveira et al. 2014; (39) Padullés Cubino et al. 2019; (40) Patitucci et al. 2011; (41) Pearse et al. 2018; (42) Pino et al. 2009; (43) Pool and Olden 2012; (44) Pouyat et al. 2015; (45) Reichert et al. 2017; (46) Ricotta et al. 2014; (47) Rocha et al. 2016; (48) Rogers et al. 2009; (49) Scott 2006; (50) Steele et al. 2014; (51) Trentanovi et al. 2013; (52) Uchida et al. 2018; (53) Ulrich et al. 2007; (54) Villegas Vallejos et al. 2016; (55) Vázquez-Reyes et al. 2017; (56) Wang et al. 2014; (57) Yang et al. 2015

TABLE 4. Urban homogenization (UH) results by spatial scale and comparison type.

Spatial scale	Evidence for UH	Mixed evidence	No evidence / evidence against UH
Single city - 15 studies (26%)			
Distance to urban	1 (100%) (27)	-	-
Land use gradient	5 (45%) (7, 23, 28, 37, 47)	3 (27%) (8, 20, 21)	3 (27%) (12, 16, 53)
Natural patches	1 (33%) (52)	1 (33%) (51)	1 (33%) (6)
Single country - 28 studies (49%)			
Entire cities	2 (66%) (3, 50)	1 (33%) (19)	-
Grid cells	1 (20%) (49)	-	4 (80%) (2, 32, 42, 43)
Land use gradient	9 (69%) (5, 11, 14, 30, 36, 41, 54, 55, 56)	3 (23%) (9, 15, 24)	1 (8%) (40)
Natural patches	3 (50%) (17, 45, 48)	-	3 (50%) (4, 22, 38)
Single urban land use	-	1 (100%) (39)	-
Multiple countries, one continent - 6 studies (11%)			
Entire cities	-	1 (100%) (46)	-
Grid cells	1 (100%) (26)	-	-
Land use gradient	2 (50%) (33, 34)	2 (50%) (18, 29)	-
Multiple continents - 8 studies (14%)			
Entire cities	1 (25%) (13)	2 (50%) (25, 57)	1 (25%) (1)
Land use gradient	1 (33%) (35)	2 (66%) (10, 44)	-
Natural patches	-	-	1 (100%) (31)

Notes: Small numerals indicate references. (1) Aronson et al. 2014; (2) Aronson et al. 2015; (3) Asmus and Rapson 2014; (4) Bertocci et al. 2017; (5) Blair and Johnson 2008; (6) Brice et al. 2017; (7) Catterall et al. 2010; (8) Deák et al. 2016; (9) Docherty et al. 2018; (10) Epp Schmidt et al. 2017; (11) Filloy et al. 2015; (12) Fischer et al. 2016; (13) Garcillán et al. 2014; (14) Hall et al. 2016; (15) Harrison et al. 2018; (16) Hartley et al. 2007; (17) Holway and Suarez 2006; (18) Horsák et al. 2013; (19) Huwer and Wittig 2013; (20) Jenerette et al. 2006; (21) Jim and Chen 2008; (22) Johnson et al. 2013; (23) Kale et al. 2018; (24) Knop 2016; (25) La Sorte et al. 2007; (26) La Sorte et al. 2014b; (27) Leveau and Leveau 2016; (28) Leveau et al 2015; (29) Lososová et al. 2012; (30) Luck and Smallbone 2011; (31) Magura et al. 2010; (32) Marchetti et al. 2006; (33) Morelli et al. 2016; (34) Murthy et al. 2016; (35) Newbold et al. 2018; (36) Nock et al. 2013; (37) Öckinger et al. 2009; (38) Oliveira et al. 2014; (39) Padullés Cubino et al. 2019; (40) Patitucci et al. 2011; (41) Pearse et al. 2018; (42) Pino et al. 2009; (43) Pool and Olden 2012; (44) Pouyat et al. 2015; (45) Reichert et al. 2017; (46) Ricotta et al. 2014; (47) Rocha et al. 2016; (48) Rogers et al. 2009; (49) Scott 2006; (50) Steele et al. 2014; (51) Trentanovi et al. 2013; (52) Uchida et al. 2018; (53) Ulrich et al. 2007; (54) Villegas Vallejos et al. 2016; (55) Vázquez-Reyes et al. 2017; (56) Wang et al. 2014; (57) Yang et al. 2015

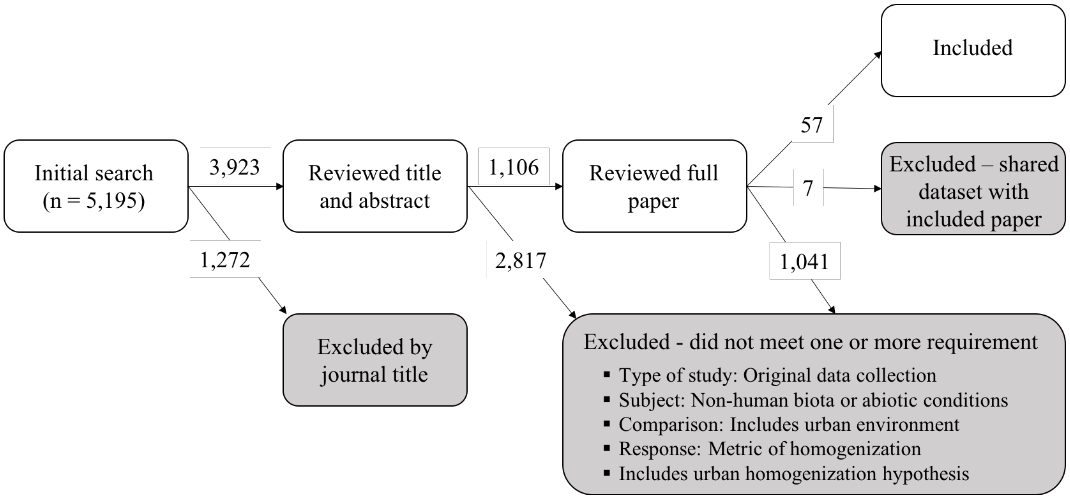

FIG. 1. Inclusion/exclusion sorting process.

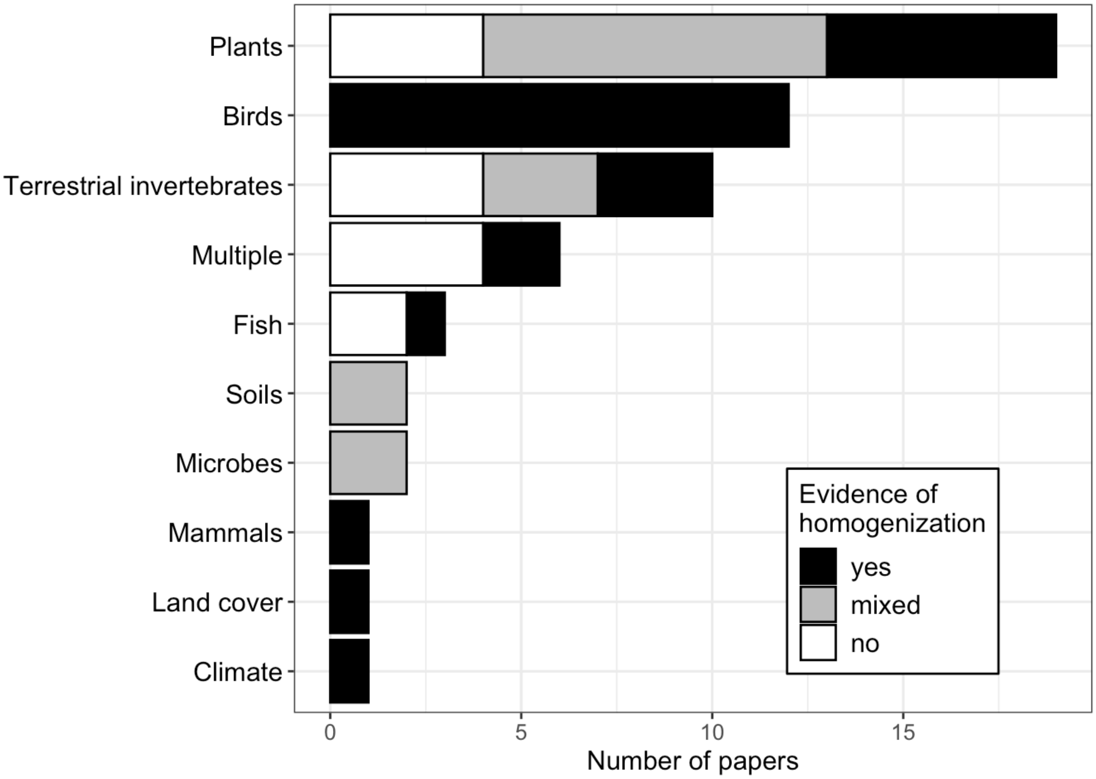

FIG. 2. Urban homogenization studies by subject, with main urban homogenization conclusion. "Multiple" indicates studies testing homogenization of multiple subjects in different groups, such as birds and plants.

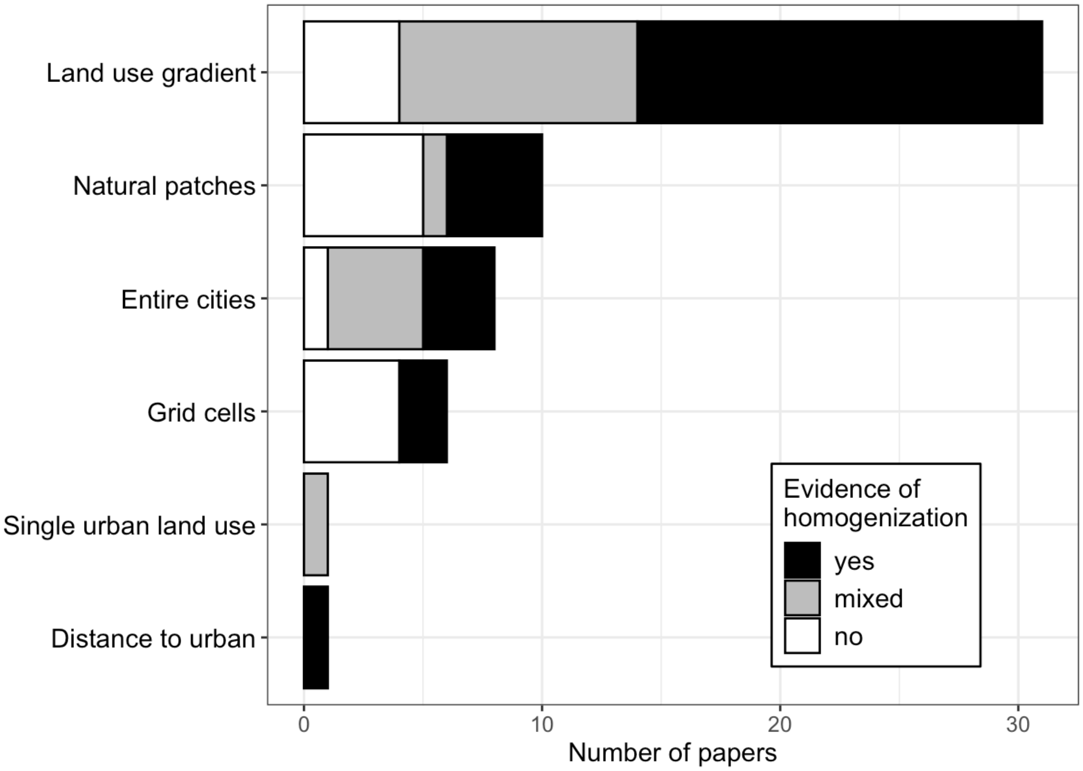

FIG. 3. Types of comparisons with which urban homogenization was evaluated, with main urban homogenization conclusion.

CHAPTER 3

WATER AND NITROGEN AVAILABILITY SHAPE DIVERSITY AND COMMUNITY COMPOSITION OF WINTER ANNUAL PLANTS IN NEAR-URBAN SONORAN DESERT PRESERVES

Abstract. Increased nitrogen (N) deposition threatens global biodiversity, but its effects in arid urban ecosystems are not well studied. In addition to altered N availability, urban environments also experience increases in other pollutants, decreased population connectivity, and altered biotic interactions, which can further impact biodiversity. In deserts, annual plant communities make up most of the plant diversity, support wildlife, and contribute to nutrient cycling and ecosystem processes. Functional tradeoffs allowing coexistence of a diversity of annual plant species are well established, but maintenance of diversity in urban conditions and with increased availability of limiting nutrients has not been explored. We conducted a 13-year N and phosphorus (P) addition experiment in Sonoran Desert preserves in and around Phoenix, Arizona, to test how nutrient availability interacts with growing season precipitation, urban location, and microhabitat to affect winter annual plant diversity. Using structural equation modeling and generalized linear mixed modeling, we found that annual plant taxonomic diversity was significantly reduced in N-enriched and urban plots. Water availability in both current and previous growing seasons impacted annual plant diversity, with significant interaction effects showing increased diversity in wetter years and greater responsiveness of the community to water following a wet year. However, there were no significant interactions between N enrichment and water availability, urban location, or microhabitat. Our results suggest that N enrichment alters the outcomes of the

physiological tradeoffs associated with water use, nutrient gathering, and growth that maintain large-scale and long-term diversity in the winter annual plant community, resulting in overall reduced taxonomic richness. Lowered diversity in urban preserves is likely partly attributable to increased urban N deposition. Changes in biodiversity of showy species like annual wildflowers in urban preserves can have important implications for connections between urban residents and nature, and reduced diversity and community restructuring with N enrichment represents a challenge for future preservation of aridland biodiversity.

INTRODUCTION

Anthropogenic activity has significantly increased global nitrogen (N) deposition and availability, which alters ecosystem structure and function (Sala et al. 2000, Bobbink et al. 2010, Pardo et al. 2011, Ackerman et al. 2019). High N availability can reduce diversity and change the species composition of primary producers with potentially long-lasting effects (Elser et al. 2007, Pardo et al. 2011, Field et al. 2014, Harpole et al. 2016, Payne et al. 2017). Urban activities, including fossil fuel burning, contribute to increased N deposition, and ecosystems within or close to cities tend to experience higher N availability than do relatively distant ecosystems (Fenn et al. 2003b, Galloway et al. 2008, Bettez and Groffman 2013). N deposition rates are expected to rise as cities grow, leading to ever greater effects on urban and surrounding regions (Fenn et al. 2003a, Liu et al. 2013, Ackerman et al. 2019). Rapid urban growth is expected in arid and semi-arid regions worldwide (Seto et al. 2011, United Nations 2014); thus, improved understanding is needed of the effects of increased N availability on biodiversity and ecological functioning in dryland regions.

Most research to date on the effects of elevated N availability on terrestrial ecosystem function has focused on grasslands and forests, with comparatively little work in arid and semi-arid ecosystems (Bobbink et al. 2010, Pardo et al. 2011). However, arid and semi-arid ecosystems make up a large fraction of the global land area (about 30%, Gamo et al. 2013) and function differently than wetter (more mesic) ecosystems in fundamental ways. In mesic systems, addition of the common limiting nutrients—N and phosphorus (P) —typically increases net primary production (NPP) while decreasing plant species diversity (Elser et al. 2007, LeBauer and Treseder 2008, Bobbink et al. 2010, Fay et al. 2015). In deserts, however, both water and nutrients limit NPP, and lack of water can diminish the effects of increased nutrient availability (Noy-Meir 1973, Hooper and Johnson 1999, Snyman 2002, Rao and Allen 2010, Yahdjian et al. 2011, Ladwig et al. 2012, Sponseller et al. 2012). These interactions between the effects of water and nutrients on NPP in arid and semi-arid systems are also likely to affect the relationships among nutrient availability, species composition, and diversity.

Desert ecosystems are inherently patchy when compared to mesic ecosystems, with a high degree of temporal and spatial heterogeneity in water availability and soil resources (Noy-Meir 1973). Rainfall is infrequent, often spatially localized, and highly variable from year to year. Sparsely distributed, long-lived shrubs and trees create "islands of fertility" under their canopies with different soil and hydrologic characteristics and increased nutrient concentrations relative to interplant spaces (Schlesinger et al. 1996, Schade and Hobbie 2005). Given this spatial and temporal variability, desert plant communities are strongly influenced by facilitative effects, whereby shrubs and trees buffer temporal variability for other species with complementary strategies for survival (Holzapfel and Mahall 1999, Butterfield and Callaway 2013, McIntire and Fajardo 2014). Thus, small-scale patchiness in deserts may lead to landscape-level plant responses to increased nutrient availability in deserts that are not observed in more mesic settings.

In addition to the variability inherent in desert ecosystems, plant communities located in aridland cities are subject to physical, chemical, and biological stressors that arise from urbanization. For example, urban greenspaces are more fragmented and experience greater air, water, noise, and light pollution than do non-urban preserves (Grimm et al. 2008, McDonald et al. 2009). Biotic homogenization, in which widespread non-native species replace local native species, has been hypothesized to lead to more similar communities across urban greenspaces than across natural preserves (McKinney 2006). Greater atmospheric deposition, and thus nutrient availability, in urban spaces may interact with desert landscape patchiness to result in different plant community responses than is observed in more natural spaces. For example, dominance of non-native species in urban communities could lead to different community-level responses to variable climatic conditions.

Annual plants are an important component of aridland plant communities and make up about 50% of all plant species diversity in the Sonoran Desert (Venable et al. 1993). These species have a rapid life cycle, which enables communities to quickly respond to yearly variations in environmental conditions (Mulroy and Rundel 1977). Desert annual plants are an important resource for pollinators and species like the threatened Agassiz's desert tortoise (*Gopherus agassizii*) (Jennings and Berry 2015) and can account for up to half of desert primary production in wet years (Hadley and Szarek 1981). Additionally, some annual plant species were an important cold-season food source for historic native societies (Bohrer 1991) and are a charismatic feature of the desert for contemporary visitors (Ryan 2011).

Much is known about how desert annual plant species respond to variability in precipitation and temperature (Venable et al. 1993, Pake and Venable 1995, Gremer et al. 2012, Huxman et al. 2013). Functional tradeoffs between water use efficiency and relative growth rate can explain long-term community coexistence among species under highly variable conditions, as species with differing strategies thrive under different sets of environmental conditions (Angert et al. 2007, Kimball et al. 2011, Gremer et al. 2013, Ge et al. 2019). Long-term monitoring of Sonoran Desert winter annuals has shown impacts of climate change on coexistence and competitive interactions, with increased abundance of cold-tolerant species (due to altered germination timing) and species with more demographically consistent populations (i.e., less dependent on yearly conditions for germination) over time (Kimball et al. 2010, Huxman et al. 2013). Following the principles of the leaf economics spectrum (Reich 2014), increased nutrient availability can alter a plant's physiological tradeoffs associated with water use, nutrient gathering, and growth, resulting in shifts in functional or phylogenetic composition. However, the co-occurring effects of chronic nutrient enrichment and climate variability have not been evaluated in these communities.

Annual plants rely entirely on the seed bank for continuity from year to year, and therefore exhibit bet-hedging strategies whereby only a small fraction of seeds germinate in any given year (Adondakis and Venable 2004, Venable 2007, Gremer and Venable 2014, Gremer et al. 2016). However, environmental conditions outside of the immediate germination and growth period can influence a given year's emergent community. For example, previous year and preceding summer conditions can affect germination response of winter annuals (Adondakis and Venable 2004, Bowers 2005), and favorable conditions for growth in a given year are likely to result in greater seed set (Pake and Venable 1995). Annual plant survival and reproduction may also depend on proximity to shrubs and their resource islands, suggesting additional complex dynamics between previous conditions, current conditions, and microsite characteristics (Pake and Venable 1995, Holzapfel and Mahall 1999). Interactions between these various drivers of annual plant composition and chronic nutrient enrichment have not been well explored, particularly over long time periods capturing both spatial and temporal variation in environmental conditions.

We conducted a long-term nutrient fertilization experiment in Sonoran Desert preserves across a precipitation gradient within and around metropolitan Phoenix, Arizona, to ask how nutrient enrichment, climate, microhabitat, and proximity to the urban environment interact to shape Sonoran Desert winter annual plant communities. We hypothesized that (1) annual plants are primarily limited by water, which will result in negligible effects of nutrient addition in dry years; (2) nutrient enrichment changes the nutrient acquisition and water use efficiency physiological tradeoffs of annual plant communities, leading to changes in diversity and composition; (3) urban preserves support lower annual plant diversity due to higher ambient levels of atmospheric N deposition; and (4) shrubs buffer resource variability for annual plants through facilitative interactions, resulting in higher annual plant diversity below shrub canopies in lower-resource years and treatments. We tested these hypotheses by measuring annual plant community composition in 15 sites during eight years of the long-term nutrient enrichment experiment. Because the desert preserve sites occur along a precipitation gradient as well as an urbanization gradient (with variable rates of N deposition), we predicted that urban unenriched communities would have similar species diversity as non-urban N-fertilized communities, and that diversity would increase along the precipitation gradient during wet years. Additionally, we predicted higher annual plant diversity in under shrub spaces, especially in dry years.

METHODS

Site description

We established 15 sites in native Sonoran Desert preserves within the Central Arizona-Phoenix Long Term Ecological Research (CAP LTER) study area, in and around the Phoenix metropolitan area (Fig. 1, Table 1). Sites were stratified by region relative to the metro Phoenix urban core, (hereafter "region"), with five sites in the west valley, five in the east valley, and five in the metropolitan area (for additional site description, see Hall et al. 2011, Cook et al. 2018). Winter precipitation varied predictably across regions, with increasing rainfall from west to east (Table 1). Measured N deposition in the urban sites (7.2 ± 0.4 kg N ha^{-1} y^{-1}) was elevated compared to the surrounding preserves to the east and west (6.1 ± 0.3 kg N ha^{-1} y^{-1}), and was lower than expected from measurements in other cities (Cook et al. 2018). To quantify the effects of region relative to the urban core, we compare annual plant communities and environmental variables in the five desert preserves in the metropolitan area (hereafter "urban") to the 10 desert preserves outside the city (hereafter "non-urban").

Nutrient enrichment treatments

At each of the 15 sites, we established four nutrient-addition treatment plots: N-fertilized, P-fertilized, N + P-fertilized, and unfertilized/control. Plots were 20 m x 20 m squares in order to capture landscape patchiness and create large fertilized areas, and plots at a site were at least 5 m apart. Nutrient treatments within a site were randomly assigned, and plots were located with consideration of topography to avoid runoff from fertilized to unfertilized plots. Each plot contained at least five individuals of three common shrubs, *Larrea tridentata* (DC.) Coville, *Ambrosia deltoidea* (Torr.) W.W. Payne, or *Ambrosia dumosa* (A. Gray) W.W. Payne. Plots excluded leguminous trees.

Nutrient treatments were added as hand-broadcast solids twice annually, once between December and February and once between June and August to follow the first winter and summer rains, respectively. Fertilization began in December of 2005. Phosphorus-enriched plots received triple superphosphate at 120 kg P ha^{-1} yr^{-1} from 2006-2008, reduced to 60 kg P ha^{-1} yr^{-1} in 2009 and 12 kg P ha^{-1} yr^{-1} from 2010-2018. Phosphorus fertilization was initially in excess to increase the probability of P reaching deep shrub roots and was decreased over time. Nitrogen in the form of ammonium nitrate (NH_4NO_3) was initially added at twice the hypothesized rate of N deposition in urban centers (Fenn et al. 2003b) and then decreased after ten years, with 60 kg N ha^{-1} yr^{-1} from 2006-2015, 45 kg N ha^{-1} yr^{-1} in 2016, and 30 kg N ha^{-1} yr^{-1} in 2017 and 2018. N + P-fertilized plots received both amendments at the rates given above.

Annual plant composition

We measured winter annual plant community composition at estimated peak biomass, which was between February and March depending on the timing of winter rainfall. Peak biomass was in February in all years except 2013, when it occurred in March. Yearly measurements began in 2008 and continued through 2018, with the exceptions of 2011, 2012, and 2014, years with relatively low winter precipitation. Low precipitation conditions were captured in 2018.

In each 20 m x 20 m treatment plot, we established four permanent 1 x 1 m quadrats for plant community composition measurements. Quadrats were located in different microhabitats: two under *Larrea tridentata* shrub canopies and two in the open spaces between shrubs (Facelli and Temby 2002, Schade and Hobbie 2005). All annual species within each quadrat were identified to the lowest possible taxon (<1% not identifiable to genus; Appendix B: Table S1). Species nomenclature follows The Plant List (2013). Species nativity to Arizona was determined from the USDA Plants (USDA NRCS 2020). Genera were defined as native if only native species were observed, non-native if only non-native species were observed, and mixed if both native and non-native species were observed or if genus-level identifications may have been either native or non-native species (Appendix B: Table S1).

The abundance of each taxon was estimated as the total fraction of the 1 m x 1 m quadrat covered by individuals of that taxon. For all years after 2008, overlapping species were counted separately, so the total cover of all species in a plot may be greater than 100%. In 2008, relative abundance of species was estimated rather than percent cover, such that the relative abundance of all species in a plot summed to 100. For this reason, abundance data for 2008 are considered separately from all other years.

We established four additional permanent 1 m x 1 m quadrats within the 20 m x 20 m plots for measurement of annual aboveground net primary productivity (ANPP) in the same years as community composition sampling. As with community composition, two quadrats were located under shrubs and two between shrubs. We clipped all aboveground living plant material at the soil surface from 0.25 m^2 subplots in the corners of permanent 1 m x 1 m quadrats. The subplot clipped rotated each year, such that there were at least three years between clipping the same subplot. Collected plant material was dried at 60°C for 48 hours then weighed to get an estimate of ANPP. Measured ANPP in the replicate quadrats was averaged to give one estimate (g m^{-2}) for each combination of site, nutrient treatment, and microhabitat per year.

Some irregularities occurred across our eight years of annual plant sampling when permanent plots could not be located or accessed in a particular year. ANPP was not sampled at SNE, SNW, DBG, MVP, MCN, or UMP in 2018; at UMP, SRR, MCS, or MVP in 2015; or at three DBG plots and one SRR plot in 2013, resulting in a total of 876 ANPP samples for all years. Additionally, plots at MVP were sampled for ANPP but not community composition in 2008. We therefore analyzed a total of 952 samples for community composition (15 sites x 4 treatments x 2 microhabitats x 8 years - 1 site x 4 treatments x 2 microhabitats x 1 year; Grimm et al. 2019).

Diversity metrics

All analyses were performed in R version 3.6.1 (R Core Team 2019). We quantified annual plant diversity using species richness, Shannon diversity, and phylogenetic diversity as metrics. For all diversity metrics, species lists from the two replicate community composition subplots were combined and measured cover was averaged by species. The most commonly observed genus, *Pectocarya*, contained three species (*P. recurvata*, *P. heterocarpa*, and *P. platycarpa*), but was reduced to genus and treated as a single species for the purpose of these analyses due to difficulty identifying to species when fruits were immature. Shannon diversity was calculated using community percent cover data for all years except 2008, when species cover was collected in a non-comparable way. We calculated species richness using the function specnumber and Shannon diversity using the function diversity from R package vegan (version 2.5.3, Oksanen et al. 2018).

Phylogenetic diversity can give additional insight into the composition of communities relative to more simplistic measures such as species richness by describing the evolutionary history and relatedness of communities (Webb et al. 2002, Cavender-Bares et al. 2009). Additionally, in the absence of physiological trait measurements, phylogenetic diversity can act as a proxy for functional diversity (Webb et al. 2002, Cavender-Bares et al. 2009). To determine phylogenetic distances between taxa, we used the angiosperm phylogeny defined by Smith and Brown (2018) and constructed from GenBank and Open Tree of Life taxa with a backbone provided by Magallón et al. (2015). This tree was chosen in place of the commonly used phylogeny from Zanne et al. (2014) because it included all genera recorded in this study. We considered phylogenetic relationships at the genus rather than species level because congeners were sometimes cryptic and difficult to distinguish in all years (e.g., *Pectocarya recurvata*, *P. heterocarpa*, and *P. platycarpa*). Multiple species were observed for 16 genera (out of 78 total), with more than two distinct species observed for only *Cryptantha* (5 species) and *Pectocarya* (3 species; Appendix B: Table S1). Comparisons at the genus rather than species level may result in higher measured phylogenetic diversity if many congenerics are present; however, most diversity in this community is captured at the genus level as there were few recorded congenerics for nearly all genera. Using the function congeneric.merge from R package pez (version 1.1-1, Pearse et al. 2015), we merged all species into the tree, reduced the entire tree to genus, and trimmed it to include only genera recorded in this study (Appendix B: Fig. S1).

Phylogenetic trees were visualized using R package ggtree (version 1.16.6, Yu et al. 2017). For the purpose of visualizations, genera observed in less than 0.5% of samples (<5 observations across all 952 community composition plots, 29 genera out of 78 total) were removed (Fig. 2; Appendix B: Fig. S1, Table S1). Trees are labeled by family to show groupings of potential functional importance. All observed genera were included in calculated diversity metrics.

We considered phylogenetic diversity using mean pairwise distance (MPD), a measure of the average evolutionary distance from a taxon in a sample to its closest relative in the sample. We chose this metric to represent divergence or relatedness of communities, and to complement our species richness metric rather than using a more highly correlated metric such as Faith's phylogenetic distance (Tucker et al. 2017). Our MPD measurements were standardized against a null model where tip labels in the phylogeny were shuffled to give a standardized effect size (SES) of mean pairwise distance (hereafter "SES MPD"), or the relatedness of a community compared to a random community drawn from the phylogeny. Samples with no plants or only one genus observed could not be included in calculations of phylogenetic diversity (74 of 952 samples were therefore excluded). SES MPD was calculated in R using the ses.mpd function from package picante (version 1.7, Kembel et al. 2010).

Water availability

We collected climate data for the winter growing season, defined as beginning in October when annual plants first germinate (Pake and Venable 1995, Venable and Pake 1999), and ending at peak biomass, when annual plant community composition was sampled each year.

To represent the overall climatic conditions in each year and site, we used a simple aridity index defined as total precipitation (mm) divided by potential evapotranspiration (PET; mm) (UNEP 1992). This aridity index (hereafter "water availability") estimates the amount of water inputs relative to PET as estimated by temperature, such that higher values indicate wetter conditions and lower values indicate drier conditions (Fig. 3). In addition to the current growing season water availability, we considered water availability in the previous growing season, defined as October through the end of March of the year before annual plant sampling, as a potentially important antecedent condition that may affect current season annual plant growth.

We gathered precipitation data from the Flood Control District of Maricopa County (FCDMC) and temperature data (used to calculate PET) from both FCDMC and the National Climate Data Center (NCDC) (Flood Control District of Maricopa County 2018, NOAA 2018). Each of our 15 study sites was matched with the nearest 3-5 precipitation stations and 1-2 temperature stations from these sources (Appendix B: Table S2). Precipitation stations were located within 10 km ground distance and 150 m elevation distance from each site if possible, while temperature stations were within 20 km ground distance and 100 m elevation distance from each site. For precipitation stations, a small ground distance was considered more important than a small elevation difference due to typically patchy rainfall patterns in the region, while for temperature stations elevation was considered more important to avoid temperature gradients with altitude. Where no data were available for a given site and day from stations within these distances, more distant stations were used (Appendix B: Table S3). For two sites (DBG and LDP), micrometeorological stations maintained by CAP LTER were located on site beginning in 2010 (Grimm et al. 2017). When and where available, these data were used in place of data from FCDMC or NCDC sensors. We averaged daily values of rainfall, maximum temperature, and minimum temperature where data from multiple stations were available for a given site and day.

To calculate PET, we input monthly average minimum and maximum temperatures to the hargreaves function in R package SPEI (version 1.7, Beguería and Vicente-Serrano 2017), using site latitudes to estimate radiation. Hargreaves PET has been shown to perform well in arid and semi-arid environments (Samani and Pessarakli 1986, Hargreaves and Allen 2003) and requires only temperature data. We then summed precipitation and predicted Hargreaves PET over the entire growing season and divided precipitation by PET to get our calculated relative water availability index (Fig. 3).

Diversity and community composition analysis

We used structural equation modeling (SEM) and generalized linear mixed modeling to evaluate the effects of nutrient addition (N and P), water availability, urban location (urban/non-urban), and microhabitat (under or between shrubs) on diversity of Sonoran Desert winter annual plants. With SEM, we explored how our various predictors were related to multiple response variables through direct and indirect pathways. With mixed modeling, we investigated interaction effects between predictors, taking into account the nested design by site. We also considered changes in community composition with these predictors using ordination, PERMANOVA, and similarity percentage (SIMPER) analyses.

We used SEM to compare the effects of our predictors on annual plant taxonomic diversity (as defined by species richness and Shannon diversity) and phylogenetic diversity (defined by SES MPD). We also tested for indirect effects of our predictors on taxonomic diversity via changes in ANPP (Fig. 4), as expected if diversity is reduced through increased light or other resource competition (Hautier et al. 2009). The SEM was fit using R package lavaan (version 0.6-5, Rosseel 2012) and plotted using package semPlot (version 1.1, Epskamp 2019). A total of 704 samples had complete data (omitted 248 samples missing SES MPD, Shannon diversity, and/or ANPP) and were used to fit the SEM. The latent variable "taxonomic diversity" was defined by species richness and Shannon diversity, with the loading for species richness set to 1. The SEM fit with Tucker-Lewis Index (TLI) = 0.83, Comparative Fit Index (CFI) = 0.96, Standardized Root Mean Square Residual (SRMR) = 0.03, and Root Mean Square Error of

Approximation (RMSEA) = 0.09. While the commonly used RMSEA metric indicates only an acceptable model fit, it is known to be overly strict in cases with few degrees of freedom such as this model with eight degrees of freedom (Kenny et al. 2015). TLI also does not indicate a good fit (<0.95), but CFI and SRMR do support the model fit (>0.95 and <0.08 respectively) (Taasoobshirazi and Wang 2016), so we accepted the model as a reasonable representation of the data.

We further explored drivers of taxonomic diversity using generalized linear mixed models, considering interaction effects between predictors and allowing random intercepts by site. We chose to use species richness as the response variable for these models, as SEM results showed that our predictors were able to explain most variation in taxonomic diversity as defined primarily by species richness. Fixed factors included in the global model were N addition (no addition as the base level), P addition (no addition as the base level), urban region (non-urban as the base level), microhabitat (location between shrubs as the base level), growing season water availability, previous growing season water availability, and all pairwise interactions. As current and previous year water availability were the only numeric predictors included and were on the same scale, they were not standardized. To accommodate our nested design, we included a random intercept for site. Models were fit with the glmer function in the lme4 package (version 1.1.19, Bates et al. 2015) using a Poisson distribution to account for count data. The overdispersion factor for the global model was 1.1 as measured using the function dispersion_glmer in package blmeco (version 1.4, Korner-Nievergelt et al. 2015), so we assume no overdispersion. All possible models were created from the global model using the dredge function in the MuMIn package (version 1.42.1, Bartoń 2018). Models within delta AICc of 2 from the top model (8 models; Appendix B: Table S4) were averaged using the model.avg function in MuMIn and considering the full average for predictor coefficient estimates (Burnham and Anderson 2004).

In addition to modeling diversity, we evaluated the relationships between annual plant community composition and nutrient treatment, urban location, microhabitat, and water availability using functions in the R package vegan (version 2.5.3, Oksanen et al. 2018). We calculated the genus-level community dissimilarity matrix using presence/absence of each genus with the binary Bray-Curtis dissimilarity index using the function vegdist. We then used the function metaMDS to create a nonmetric multidimensional scaling (NMDS) plot, allowing three dimensions, to visualize differences in community composition by our variables of interest. To determine statistical significance of changes in community composition, we fitted environmental vectors for current and previous season water availability to the NMDS using function envfit, with fit significance determined by permutation tests with 999 permutations. We tested for effects of urban location, nutrient treatment, and microhabitat on community composition using PERMANOVA, followed by a similarity percentage (SIMPER) analysis to identify discriminating genera between group levels (e.g., urban/non-urban, N/no N). These analyses were run with the adonis and simper functions, respectively. For PERMANOVA, permutations were restricted by site to account for the nested design, and 999 permutations were run. Significant SIMPER differences in genera by urban location, nutrient treatment, and microhabitat were determined using permutation tests with 100 permutations. We also checked for differences in within-group variance using function betadisper.

RESULTS

Both structural equation modeling and mixed modeling approaches showed negative impacts of N addition and urban location on annual plant taxonomic diversity (Fig. 5, Table 2). Current and previous season water availability were significant predictors of taxonomic diversity with complex interaction effects. The effect of microhabitat on taxonomic diversity was significant and positive (i.e., greater diversity under shrubs) with SEM (Fig. 5), although mixed modeling did not find a significant effect of microhabitat on species richness (Table 2). While nutrients, urban location, and water directly impacted productivity, there was no evidence of indirect effects of nutrients, water, microhabitat, or urban location on taxonomic diversity via productivity (Appendix B: Table S5). Only microhabitat significantly affected phylogenetic diversity. Overall community composition shifted significantly in response to N, P, water, microhabitat, and urban location, although P did not affect either taxonomic or phylogenetic diversity. In the following sections, we investigate these relationships with respect to our hypotheses.

Effects of nutrients and water

N enrichment significantly reduced annual plant taxonomic diversity but not phylogenetic diversity, while P enrichment was not significantly related to either taxonomic or phylogenetic diversity (Fig. 5). No significant interaction was found between N addition and other environmental variables. The lack of significant interactions suggests that N addition suppressed annual plant diversity in both wet and dry conditions (Fig. 6), regardless of urban location or microhabitat. Community composition was significantly different with both N enrichment (PERMANOVA, partial $R^2 = 0.018$, $p = 0.001$; Appendix B: Fig. S2) and P enrichment (PERMANOVA, partial $R^2 = 0.004$, $p = 0.001$; Appendix B: Fig. S3) when accounting for differences by site. Nearly all genera were less common in N-enriched plots (Fig. 7b), with *Plantago*, *Logfia*, *Erodium*, and *Vulpia* having the largest significant contributions to community differentiation by N enrichment (Fig. 8a). All Fabaceae genera were less commonly found in N-fertilized plots (Fig. 7b). Additionally, N-enriched plots were more similar in composition to one another than were plots without N (betadisper, $p = 0.02$). Responses to P enrichment were more varied (Fig. 7c).

While SEM showed a positive effect of current season water availability and a negative effect of previous season water availability on annual plant taxonomic diversity, mixed modeling of species richness included significant interaction effects between urban location, current growing season water availability, and previous growing season water availability (Table 2, Fig. 9). With mixed modeling, the main effects of both current and previous season water availability on species richness were negative, but there was also a large, positive interaction effect between current and previous season water availability (Fig. 9). Annual plant species richness increased with current season water availability more following a wetter year than after a dry year, with little effect of current season water availability on species richness following a dry year. Species richness responded more positively to both current and previous season water availability in urban than in non-urban sites (Fig. 9). However, a broader range of water availability conditions was measured in non-urban sites (current season 0.04 - 0.79; previous season 0.05 - 0.59) than in urban sites (current season 0.05 - 0.42; previous season 0.08 - 0.30), and thus, predictions for urban environments outside the measured range should be interpreted with caution. Additionally, due to dry conditions in March, water availability (precipitation/PET) for a given year was often lower when considered as the previous year's water availability (Oct-Mar) rather than the current season water availability (Oct-Feb, except 2013).

Following ordination analysis, both current and previous growing season water availability significantly distinguished community composition across samples (Fig. 10). However, phylogenetic diversity did not vary significantly with water availability (Fig. 5).

Effects of urban location

Urban sites had significantly lower taxonomic diversity than did non-urban sites, and species richness was more responsive to changes in water availability in urban than in non-urban sites (Fig. 5, Fig. 9, Table 2). Community composition differed between urban and non-urban locations when accounting for site (PERMANOVA, partial R^2 = 0.047, p = 0.001; Appendix B: Fig. S4), and there was less variation in community composition in urban than in non-urban sites (i.e., urban sites had more similar community composition to one another than did non-urban sites; betadisper, p = 0.03). Differentiation between urban and non-urban sites was most strongly driven by eight taxa that were less common in urban sites (*Amsinkia*, *Plantago*, *Lepidium*, *Logfia*, *Erodium*, *Crassula, Pectocarya*, and *Schismus*) and three that were more common in urban sites (*Cryptantha*, *Draba*, and *Euphorbia*; Fig. 7, Fig. 8). At the species level, *Cryptantha* exhibited varied responses to urban location, with *C. maritima* more commonly found in urban sites, *C. angustifolia* more commonly found in non-urban sites, and the most frequently observed species, *C. decipiens*, nearly evenly distributed between urban and non-urban sites. Few congenerics were observed for genera other than *Cryptantha*, and no other highly differentiating genus contained species with different responses to urban location. As observed with N addition, most genera were less common in urban sites. The non-native or partially non-native genera *Plantago*, *Schismus,* and *Erodium* all significantly differentiated urban and non-urban sites, but were all more common in non-urban locations (Fig. 7, Fig. 8).

Effects of microhabitat

Taxonomic diversity was higher beneath shrubs while phylogenetic diversity was reduced (i.e., more taxonomically diverse but phylogenetically clustered communities under shrubs; Fig. 5). Microhabitat was the only significant predictor of phylogenetic diversity. Community composition differed significantly by microhabitat (PERMANOVA, partial $R^2 = 0.024$, $p = 0.001$) when accounting for site (Appendix B: Fig. S5). SIMPER analysis identified *Amsinckia* as the top genus contributing to differences by microhabitat, followed by *Cryptantha*, *Plantago*, and *Lepidium* (Fig. 8). *Amsinckia* and *Cryptantha*, both members of the Boraginaceae family, were more commonly found under shrubs (Fig. 7). Most grasses (family Poaceae) and mustards (family Brassicaceae, including *Lepidium*) were also more common under shrubs, while members of Fabaceae, Asteraceae, and several other families that were less diverse in this community (including *Plantago*, family Plantaginaceae) were more often found between shrubs (Fig. 7).

DISCUSSION

Maintenance of native biodiversity in the context of rapid urbanization and changing climate patterns is a major challenge for conservation. With our long-term experimental approach, we found that annual plant communities in Sonoran Desert preserves had reduced diversity in the urban core and in experimentally N-enriched plots compared to unenriched and non-urban locations. Diversity was impacted by water availability in both the current and previous growing season, with interactions showing evidence of multi-year effects of water availability on annual plants. The decline in diversity with N addition was not moderated by water availability or microhabitat. Our use of multiple statistical techniques and community diversity metrics allows us to more fully elucidate these interactions and describe their effects. Our findings suggest that arid and semi-arid annual plant biodiversity is likely to decline with increased N deposition, despite water limitation and the marked spatial and temporal resource patchiness that characterizes desert environments.

Water and nutrient addition effects on annual plant communities

Experimental N addition had uniformly negative effects on annual plant taxonomic diversity. Meanwhile, P addition had no effect on annual plant diversity. Our observed N addition effect matches previous findings in more mesic systems showing declines in plant diversity with increased N (Pardo et al. 2011, Harpole et al. 2016, Payne et al. 2017). However, contrary to our predictions, the effects of N on diversity were not moderated by water availability or microhabitat as expected if water served as the primary driver followed by N. Rather, we found support for co-regulation of winter annual plant composition and diversity by water and N.

N enrichment leading to greater biomass of dominant plant species and thus reducing overall diversity due to light limitation and outcompetition is a typical mechanism for declining diversity with fertilization (Hautier et al. 2009, Borer et al. 2014). Yet, we found no relationship between productivity and diversity that would support this interpretation in our study. Previous research at these sites also showed no relationship between annual plant ANPP and diversity following herbivore exclusion (Davis et al. 2015). The absence of primary limitation by water further suggests a different mechanism for diversity loss following N enrichment. We suggest that N decreases diversity in this community by altering the success of different physiological tradeoffs, such as water use efficiency and growth rate strategies, that maintain long-term coexistence under altered nutrient conditions regardless of water or microhabitat (Angert et al. 2009, Gremer et al. 2013, Huxman et al. 2013, Ge et al. 2019).

Nitrogen allocation is fundamental to the tradeoff between relative growth rate (RGR) and water use efficiency (WUE), which is a key tradeoff in desert plant communities (Angert et al. 2007, Gremer et al. 2013, Huxman et al. 2013). In this context, introduction of non-native species and long-term changes in climate have been shown to alter RGR and WUE strategies, with resulting changes to the composition of the overall winter annual plant community (Gremer et al. 2012, Huxman et al. 2013, Kimball et al. 2014). Increased N deposition beyond the critical N load for this community is likely to affect fitness and interactions of species along the RGR-WUE tradeoff gradient as well. While we did not observe changes in phylogenetic clustering of communities with N enrichment or urban location, previous studies have shown that annual plant species of the same genus can differ in their position along this tradeoff gradient (Huxman et al. 2008). Accordingly, a change in taxonomic diversity due to promotion of different resource allocation strategies would not necessarily correspond to a change in phylogenetic diversity. Comparisons of individual species responses along the RGR-WUE continuum to N fertilization and urban location would yield additional insight into how coexistence patterns are affected by N availability.

Our findings illustrate a complex relationship between water availability and annual plant diversity that is likely mediated through the seed bank. The interacting effects of current water availability and water availability in the previous growing season on annual plant diversity may be due to increased seed production in wet years. Therefore, we see greater responsiveness of the community to water following wet than following dry years. In this community, germination, growth, and reproduction of some species are more responsive to variation in environmental conditions, such as timing of precipitation and temperature variability, while other species have more consistent germination and growth patterns across years (Venable 2007, Angert et al. 2010, Huxman et al. 2013). Annual plant seeds have variable, low germination rates and may lie dormant in the seed bank for long periods of time (Adondakis and Venable 2004), but most of the viable seeds in a particular year are likely to have been produced in the previous year (Moriuchi et al. 2000). Thus, although the seed bank is able to preserve diversity over the long term, the majority of annual plants in a given year are likely to result from seeds in the previous year and are thereby influenced by previous year growing conditions.

Differences in responsiveness of species to yearly conditions could accumulate over time. For example, multiple wet years produce a rich seed bank of species ready to respond to further wet conditions, whereas dry years produce dominant species' seeds that may not be as responsive to wet conditions. Although we show a relationship between the previous year's environmental conditions and the current year's annual plant responses, long-term shifts in climate, including rainfall patterns, could lead to longer-term changes in the seed bank and the resulting annual plant community (Huxman et al. 2013). While we present here two possible mechanisms for changes in annual plant diversity in response to nutrient addition and water availability (changes in RGR-WUE tradeoff strategies and alterations in the seed bank over time), further research is needed to explore and confirm these mechanisms.

One limitation to this study is the consideration of both water availability and annual plant community responses at the yearly time scale. Previous work has shown that winter annual plant species respond differently to the timing of rainfall and temperature conditions within the winter growing season, and that species dominance may change over the course of the season (Kimball et al. 2012, Huxman et al. 2013). Repeated sampling during the growing season to determine community responses to individual rainfall events and temperature changes in urban and N-enriched conditions would complement and extend the present analysis by adding insight into intra-annual variability and temporal as well as spatial heterogeneity.

Diversity in urban deserts

Annual plant diversity was lower and community composition was altered in urban Sonoran Desert preserves within the city compared to non-urban preserves at the outer edges of the city. Although N deposition is relatively low in these sites compared to other cities (Cook et al. 2018), the response of our community to urban location and N enrichment suggests that even the relatively low N deposition in these urban sites is above the critical N threshold for this system (Pardo et al. 2011). Other urban conditions such as increased ozone pollution, park use by people and pets, and population isolation may also affect plant diversity, although previous research shows that changes in food web structure (e.g., increased herbivory due to loss of predators) in urban preserves would not result in the observed decrease in annual plant diversity (Davis et al. 2015). In our study, even already N-enriched urban locations experienced reductions in annual plant diversity with further experimental N addition, indicating that continued losses of diversity are likely if N deposition increases. Although urban open space parks can be managed to preserve intact soils and woody plant structure, ephemeral desert plant communities with rapid life cycles may be difficult to maintain when the negative impacts of nearby air pollution cross preserve boundaries. Nonetheless, urban preserves provide valuable access to natural landscapes for city residents and can provide refugia for native wildlife (Cox et al. 2018, Threlfall and Kendal 2018). Maintaining diverse plant communities—even in these relatively disturbed spaces—is a desirable goal for city managers and residents.

Despite this management goal, we observed greater similarity among urban annual plant communities than among non-urban communities, as expected following the urban homogenization hypothesis (McKinney 2006). However, we did not find more non-natives in urban preserves compared to non-urban preserves, suggesting that an increase in non-native species did not drive the observed homogenization pattern. Additionally, non-native annual plants were not more frequently found in N-enriched plots, contrary to previous findings of increased non-native grass dominance in N-fertilized desert sites (Brooks 2003, Rao and Allen 2010). Rather, it appears that native species were lost in urban locations and with N addition, but were not replaced by non-natives, leading to some of the observed declines in overall diversity. Other work in the Sonoran Desert has shown increased non-native annual forb and grass frequency over a longer time period (1983 - 2005) at a single site near Tucson (Bowers et al. 2006); in our sites, an increase in non-native species is not likely due to elevated N deposition or other urban influences. All of the preserves in our study as well as the site studied by Bowers et al. (2006) have experienced some disturbance related to road construction, recreation, and/or scientific research and thus may all share some conditions allowing for non-native expansion, even for those preserves we designate as non-urban. Non-native annual plant spread in this community may be more related to low-level disturbance than to highly urban conditions.

Landscape patchiness and the effects of microhabitat

We found mixed evidence for increased annual plant diversity under shrubs, which we would expect from the existence of fertile resource islands that accumulate under their canopies (Schlesinger and Pilmanis 1998, Facelli and Temby 2002). Our SEM approach identified a small positive relationship between location under shrubs and annual plant diversity, suggesting facilitation of annual plants by shrubs. Multi-model inference from mixed modeling included microhabitat in all eight top models, though it was not identified as a significant predictor. Considering these combined approaches and previous research on this topic, we conclude that shrubs do promote greater annual plant taxonomic diversity. However, species under and between shrubs had similar responses to water availability, nutrient enrichment, and urban conditions, unlike the differential responses under and between shrubs observed in other studies (Pake and Venable 1995, Brooks 2003). The benefits of facilitation by shrubs for winter annual plants in this community may be relatively small compared to the range of responses induced by interannual climatic variability, and thus unlikely to buffer the effects of increased variability with climate change and elevated N deposition in urban areas. Additionally, shrub resource islands may have within-year temporal effects on annual plant community composition by altering germination and senescence timing (Kimball et al. 2011), which our sampling approach would not have captured. However, along with intra-annual weather variability, these temporal changes could yield additional insights into how urban and N-enrichment effects alter annual plant community composition.

In addition to differences in taxonomic diversity, we found greater phylogenetic clustering in communities under shrubs. Closely-related groups of taxa may share functional traits allowing them to benefit from conditions beneath shrub canopies (Aguilera et al. 1999, Facelli and Temby 2002), although some important traits may vary even within closely-related groups (Huxman et al. 2008). The balance between facilitation and competition is dependent on compatibility between plant functional traits, and so this more closely related group of taxa beneath shrub canopies likely represents those with functional traits that best align with conditions provided by shrubs (Butterfield and Callaway 2013).

Implications for desert conservation

How might we expect desert annual plant communities to change in the future? With the combined impacts of climate change and urban growth, annual plant communities are likely to decrease in overall diversity and experience shifts in community composition (Kimball et al. 2010, Huxman et al. 2013). Increased N deposition is a concern for maintaining long-term diversity in this ephemeral community, especially in urban preserves where admiration and enjoyment of these attractive and short-lived plants may help build an appreciation of the desert for residents of arid cities. Depletion of annual plant diversity may result in muted responses of the community to certain environmental conditions, potentially leading to even greater interannual variability in the emergent community as plant strategies adapted to some conditions become less common (Huxman et al. 2013, Bharath et al. 2020). Changes in diversity of annual plants can have important impacts on showy wildflower displays appreciated (and monetized) by people, as well as floral and herbaceous resources for desert pollinators and other wildlife (Ryan 2011, Jennings and Berry 2015). As urban populations increasingly experience a loss of connection with nature (Soga and Gaston 2016), diversity of showy species like wildflowers in accessible urban preserves may be particularly influential for building positive attitudes toward the environment. Multi-year experiments such as this one show responses to a range of environmental conditions and help predict how communities will change in the future. Our finding that annual diversity and community composition are strongly influenced by multiple years of water availability suggests that considering only the conditions of a single growing season will not be sufficient to understand and predict community outcomes. Increased variability and multi-year drought may have compounding effects on annual plant diversity, with attendant outcomes for people and wildlife.

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REFERENCES

Ackerman, D., D. B. Millet, and X. Chen. 2019. Global estimates of inorganic nitrogen deposition across four decades. Global Biogeochemical Cycles 33:100–107.

Adondakis, S., and D. L. Venable. 2004. Dormancy and germination in a guild of Sonoran Desert annuals. Ecology 85:2582–2590.

Aguilera, L. E., J. R. Gutiérrez, and P. L. Meserve. 1999. Variation in soil micro-organisms and nutrients underneath and outside the canopy of *Adesmia bedwellii* (Papilionaceae) shrubs in arid coastal Chile following drought and above average rainfall. Journal of Arid Environments 42:61–70.

Angert, A. L., J. L. Horst, T. E. Huxman, and D. L. Venable. 2010. Phenotypic plasticity and precipitation response in Sonoran Desert winter annuals. American Journal of Botany 97:405–411.

Angert, A. L., T. E. Huxman, G. A. Barron-Gafford, K. L. Gerst, and D. L. Venable. 2007. Linking growth strategies to long-term population dynamics in a guild of desert annuals. Journal of Ecology 95:321–331.

Angert, A. L., T. E. Huxman, P. Chesson, and D. L. Venable. 2009. Functional tradeoffs determine species coexistence via the storage effect. Proceedings of the National Academy of Sciences USA 106:11641–11645.

Bartoń, K. 2018. MuMIn: multi-model inference. R package version 1.43.6. https://CRAN.R-project.org/package=MuMIn

Bates, D., M. Mächler, B. Bolker, and S. Walker. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67:1–48.

Beguería, S., and S. M. Vicente-Serrano. 2017. SPEI: calculation of the standardised precipitation-evapotranspiration index. R package version 1.7. https://CRAN.R-project.org/package=SPEI

Bettez, N. D., and P. M. Groffman. 2013. Nitrogen deposition in and near an urban ecosystem. Environmental Science & Technology 47:6047–6051.

Bharath, S., E. T. Borer, L. A. Biederman, D. M. Blumenthal, P. A. Fay, L. A. Gherardi, J. M. H. Knops, A. D. B. Leakey, L. Yahdjian, and E. W. Seabloom. 2020. Nutrient addition increases grassland sensitivity to droughts. Ecology 101:e02981.

Bobbink, R., K. Hicks, J. Galloway, T. Spranger, R. Alkemade, M. Ashmore, M. Bustamante, S. Cinderby, E. Davidson, F. Dentener, B. Emmett, J.-W. Erisman, M. Fenn, F. Gilliam, A. Nordin, L. Pardo, and W. De Vries. 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. Ecological Applications 20:30–59.

Bohrer, V. L. 1991. Recently recognized cultivated and encouraged plants among the Hohokam. Kiva 56:227–235.

Borer, E. T., E. W. Seabloom, D. S. Gruner, W. S. Harpole, H. Hillebrand, E. M. Lind, P. B. Adler, J. Alberti, T. M. Anderson, J. D. Bakker, L. Biederman, D. Blumenthal, C. S. Brown, L. A. Brudvig, Y. M. Buckley, M. Cadotte, C. Chu, E. E. Cleland, M. J. Crawley, P. Daleo, E. I. Damschen, K. F. Davies, N. M. DeCrappeo, G. Du, J. Firn, Y. Hautier, R. W. Heckman, A. Hector, J. HilleRisLambers, O. Iribarne, J. A. Klein, J. M. H. Knops, K. J. La Pierre, A. D. B. Leakey, W. Li, A. S. MacDougall, R. L. McCulley, B. A. Melbourne, C. E. Mitchell, J. L. Moore, B. Mortensen, L. R.

O'Halloran, J. L. Orrock, J. Pascual, S. M. Prober, D. A. Pyke, A. C. Risch, M. Schuetz, M. D. Smith, C. J. Stevens, L. L. Sullivan, R. J. Williams, P. D. Wragg, J. P. Wright, and L. H. Yang. 2014. Herbivores and nutrients control grassland plant diversity via light limitation. Nature 508:517–520.

Bowers, J. E. 2005. El Niño and displays of spring-flowering annuals in the Mojave and Sonoran deserts. The Journal of the Torrey Botanical Society 132:38–49.

Bowers, J. E., T. M. Bean, and R. M. Turner. 2006. Two decades of change in distribution of exotic plants at the Desert Laboratory, Tucson, Arizona. Madroño 53:252–263.

Brooks, M. L. 2003. Effects of increased soil nitrogen on the dominance of alien annual plants in the Mojave Desert. Journal of Applied Ecology 40:344–353.

Burnham, K. P., and D. R. Anderson. 2004. Multimodel inference: Understanding AIC and BIC in model selection. Sociological Methods & Research 33:261–304.

Butterfield, B. J., and R. M. Callaway. 2013. A functional comparative approach to facilitation and its context dependence. Functional Ecology 27:907–917.

Cavender-Bares, J., K. H. Kozak, P. V. A. Fine, and S. W. Kembel. 2009. The merging of community ecology and phylogenetic biology. Ecology Letters 12:693–715.

Cook, E. M., R. Sponseller, N. B. Grimm, and S. J. Hall. 2018. Mixed method approach to assess atmospheric nitrogen deposition in arid and semi-arid ecosystems. Environmental Pollution 239:617–630.

Cox, D. T. C., D. F. Shanahan, H. L. Hudson, R. A. Fuller, and K. J. Gaston. 2018. The impact of urbanisation on nature dose and the implications for human health. Landscape and Urban Planning 179:72–80.

Davis, M. K., E. M. Cook, S. L. Collins, and S. J. Hall. 2015. Top-down vs. bottom-up regulation of herbaceous primary production and composition in an arid, urbanizing ecosystem. Journal of Arid Environments 116:103–114.

Elser, J. J., M. E. S. Bracken, E. E. Cleland, D. S. Gruner, W. S. Harpole, H. Hillebrand, J. T. Ngai, E. W. Seabloom, J. B. Shurin, and J. E. Smith. 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecology Letters 10:1135–1142.

Epskamp, S. 2019. semPlot: path diagrams and visual analysis of various SEM packages' output. R package version 1.1.2. https://CRAN.R-project.org/package=semPlot

Facelli, J. M., and A. M. Temby. 2002. Multiple effects of shrubs on annual plant communities in arid lands of South Australia. Austral Ecology 27:422–432.

Fay, P. A., S. M. Prober, W. S. Harpole, J. M. H. Knops, J. D. Bakker, E. T. Borer, E. M. Lind, A. S. MacDougall, E. W. Seabloom, P. D. Wragg, P. B. Adler, D. M. Blumenthal, Y. M. Buckley, C. Chu, E. E. Cleland, S. L. Collins, K. F. Davies, G. Du, X. Feng, J. Firn, D. S. Gruner, N. Hagenah, Y. Hautier, R. W. Heckman, V. L. Jin, K. P. Kirkman, J. Klein, L. M. Ladwig, Q. Li, R. L. McCulley, B. A. Melbourne, C. E. Mitchell, J. L. Moore, J. W. Morgan, A. C. Risch, M. Schütz, C. J. Stevens, D. A. Wedin, and L. H. Yang. 2015. Grassland productivity limited by multiple nutrients. Nature Plants 1:15080.

Fenn, M. E., J. S. Baron, E. B. Allen, H. M. Rueth, K. R. Nydick, L. Geiser, W. D. Bowman, J. O. Sickman, T. Meixner, D. W. Johnson, and P. Neitlich. 2003a. Ecological effects of nitrogen deposition in the western United States. BioScience 53:404–420.

Fenn, M. E., R. Haeuber, G. S. Tonnesen, J. S. Baron, S. Grossman-Clarke, D. Hope, D. A. Jaffe, S. Copeland, L. Geiser, H. M. Rueth, and J. O. Sickman. 2003b. Nitrogen emissions, deposition, and monitoring in the western United States. BioScience 53:391–403.

Field, C. D., N. B. Dise, R. J. Payne, A. J. Britton, B. A. Emmett, R. C. Helliwell, S. Hughes, L. Jones, S. Lees, J. R. Leake, I. D. Leith, G. K. Phoenix, S. A. Power, L. J. Sheppard, G. E. Southon, C. J. Stevens, and S. J. M. Caporn. 2014. The role of nitrogen deposition in widespread plant community change across semi-natural habitats. Ecosystems 17:864–877.

Flood Control District of Maricopa County. 2018. Weather sensor data. https://www.maricopa.gov/3769/Weather-Sensor-Data.

Galloway, J. N., A. R. Townsend, J. W. Erisman, M. Bekunda, Z. Cai, J. R. Freney, L. A. Martinelli, S. P. Seitzinger, and M. A. Sutton. 2008. Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. Science 320:889–892.

Gamo, M., M. Shinoda, and T. Maeda. 2013. Classification of arid lands, including soil degradation and irrigated areas, based on vegetation and aridity indices. International Journal of Remote Sensing 34:6701–6722.

Ge, X.-Y. M., J. P. Scholl, U. Basinger, T. E. Huxman, and D. L. Venable. 2019. Functional trait trade-off and species abundance: insights from a multi-decadal study. Ecology Letters 22:583–592.

Gremer, J. R., S. Kimball, A. L. Angert, D. L. Venable, and T. E. Huxman. 2012. Variation in photosynthetic response to temperature in a guild of winter annual plants. Ecology 93:2693–2704.

Gremer, J. R., S. Kimball, K. R. Keck, T. E. Huxman, A. L. Angert, and D. L. Venable. 2013. Water-use efficiency and relative growth rate mediate competitive interactions in Sonoran Desert winter annual plants. American Journal of Botany 100:2009–2015.

Gremer, J. R., S. Kimball, and D. L. Venable. 2016. Within-and among-year germination in Sonoran Desert winter annuals: bet hedging and predictive germination in a variable environment. Ecology Letters 19:1209–1218.

Gremer, J. R., and D. L. Venable. 2014. Bet hedging in desert winter annual plants: optimal germination strategies in a variable environment. Ecology Letters 17:380–387.

Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai, and J. M. Briggs. 2008. Global Change and the Ecology of Cities. Science 319:756–760.

Grimm, N. B., S. J. Hall, J. P. Kaye, J. Allen, and D. L. Childers. 2017. CAP weather stations at Papago Park and Lost Dutchman State Park in the greater Phoenix metropolitan area, ongoing since 2010. https://sustainability.asu.edu/caplter/data/data-catalog/view/knb-lter-cap.636/.

Grimm, N. B., S. J. Hall, J. P. Kaye, J. Allen, and D. L. Childers. 2019. Desert fertilization experiment: investigation of Sonoran Desert ecosystem response to atmospheric deposition and experimental nutrient addition, ongoing since 2006. https://sustainability.asu.edu/caplter/data/data-catalog/view/knb-lter-cap.632/.

Hadley, N. F., and S. R. Szarek. 1981. Productivity of desert ecosystems. BioScience 31:747–753.

Hall, S. J., R. A. Sponseller, N. B. Grimm, D. Huber, J. P. Kaye, C. Clark, and S. L. Collins. 2011. Ecosystem response to nutrient enrichment across an urban airshed in the Sonoran Desert. Ecological Applications 21:640–660.

Hargreaves, G. H., and R. G. Allen. 2003. History and evaluation of Hargreaves evapotranspiration equation. Journal of Irrigation and Drainage Engineering 129:53–63.

Harpole, W. S., L. L. Sullivan, E. M. Lind, J. Firn, P. B. Adler, E. T. Borer, J. Chase, P. A. Fay, Y. Hautier, H. Hillebrand, A. S. MacDougall, E. W. Seabloom, R. Williams, J. D. Bakker, M. W. Cadotte, E. J. Chaneton, C. Chu, E. E. Cleland, C. D’Antonio, K. F. Davies, D. S. Gruner, N. Hagenah, K. Kirkman, J. M. H. Knops, K. J. La Pierre, R. L. McCulley, J. L. Moore, J. W. Morgan, S. M. Prober, A. C. Risch, M. Schuetz, C. J.

Stevens, and P. D. Wragg. 2016. Addition of multiple limiting resources reduces grassland diversity. Nature 537:93–96.

Hautier, Y., P. A. Niklaus, and A. Hector. 2009. Competition for light causes plant biodiversity loss after eutrophication. Science 324:636–638.

Holzapfel, C., and B. E. Mahall. 1999. Bidirectional facilitation and interference between shrubs and annuals in the Mojave Desert. Ecology 80:1747–1761.

Hooper, D. U., and L. Johnson. 1999. Nitrogen limitation in dryland ecosystems: responses to geographical and temporal variation in precipitation. Biogeochemistry 46:247–293.

Huxman, T. E., G. Barron-Gafford, K. L. Gerst, A. L. Angert, A. P. Tyler, and D. L. Venable. 2008. Photosynthetic resource-use efficiency and demographic variability in desert winter annual plants. Ecology 89:1554–1563.

Huxman, T. E., S. Kimball, A. L. Angert, J. R. Gremer, G. A. Barron-Gafford, and D. L. Venable. 2013. Understanding past, contemporary, and future dynamics of plants, populations, and communities using Sonoran Desert winter annuals. American Journal of Botany 100:1369–1380.

Jennings, W. B., and K. H. Berry. 2015. Desert tortoises (*Gopherus agassizii*) are selective herbivores that track the flowering phenology of their preferred food plants. PLoS ONE 10:e0116716.

Kembel, S. W., P. D. Cowan, M. R. Helmus, W. K. Cornwell, H. Morlon, D. D. Ackerly, S. P. Blomberg, and C. O. Webb. 2010. Picante: R tools for integrating phylogenies and ecology. Bioinformatics 26:1463–1464.

Kenny, D. A., B. Kaniskan, and D. B. McCoach. 2015. The performance of RMSEA in models with small degrees of freedom. Sociological Methods & Research 44:486–507.

Kimball, S., A. L. Angert, T. E. Huxman, and D. L. Venable. 2010. Contemporary climate change in the Sonoran Desert favors cold-adapted species. Global Change Biology 16:1555–1565.

Kimball, S., A. L. Angert, T. E. Huxman, and D. L. Venable. 2011. Differences in the timing of germination and reproduction relate to growth physiology and population dynamics of Sonoran Desert winter annuals. American Journal of Botany 98:1773–1781.

Kimball, S., J. R. Gremer, A. L. Angert, T. E. Huxman, and D. L. Venable. 2012. Fitness and physiology in a variable environment. Oecologia 169:319–329.

Kimball, S., J. R. Gremer, G. A. Barron-Gafford, A. L. Angert, T. E. Huxman, and D. L. Venable. 2014. High water-use efficiency and growth contribute to success of non-native *Erodium cicutarium* in a Sonoran Desert winter annual community. Conservation Physiology 2.

Korner-Nievergelt, F., T. Roth, S. von Felten, J. Guelat, B. Almasi, and P. Korner-Nievergelt. 2015. blmeco: data files and functions accompanying the book "Bayesian data analysis in ecology using linear models with R, BUGS and Stan." R package version 1.4. https://CRAN.R-project.org/package=blmeco

Ladwig, L. M., S. L. Collins, A. L. Swann, Y. Xia, M. F. Allen, and E. B. Allen. 2012. Above- and belowground responses to nitrogen addition in a Chihuahuan Desert grassland. Oecologia 169:177–185.

LeBauer, D. S., and K. K. Treseder. 2008. Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. Ecology 89:371–379.

Liu, X., Y. Zhang, W. Han, A. Tang, J. Shen, Z. Cui, P. Vitousek, J. W. Erisman, K. Goulding, P. Christie, A. Fangmeier, and F. Zhang. 2013. Enhanced nitrogen deposition over China. Nature 494:459–462.

Magallón, S., S. Gómez-Acevedo, L. L. Sánchez-Reyes, and T. Hernández-Hernández. 2015. A metacalibrated time-tree documents the early rise of flowering plant phylogenetic diversity. New Phytologist 207:437–453.

McDonald, R. I., R. T. T. Forman, P. Kareiva, R. Neugarten, D. Salzer, and J. Fisher. 2009. Urban effects, distance, and protected areas in an urbanizing world. Landscape and Urban Planning 93:63–75.

McIntire, E. J. B., and A. Fajardo. 2014. Facilitation as a ubiquitous driver of biodiversity. New Phytologist 201:403–416.

McKinney, M. L. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127:247–260.

Moriuchi, K. S., D. L. Venable, C. E. Pake, and T. Lange. 2000. Direct measurement of the seed bank age structure of a Sonoran Desert annual plant. Ecology 81:1133–1138.

Mulroy, T. W., and P. W. Rundel. 1977. Annual plants: adaptations to desert environments. BioScience 27:109–114.

NOAA. 2018. Climate data online. National Climate Data Center. https://www.ncdc.noaa.gov/cdo-web/.

Noy-Meir, I. 1973. Desert ecosystems: environment and producers. Annual Review of Ecology and Systematics 4:25–51.

Oksanen, J., F. G. Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. H. H. Stevens, E. Szoecs, and H. Wagner. 2018. vegan: community ecology package. R package version 2.5-3. https://cran.r-project.org/package=vegan

Pake, C. E., and D. L. Venable. 1995. Is coexistence of Sonoran Desert annuals mediated by temporal variability in reproductive success? Ecology 76:246–261.

Pardo, L. H., M. E. Fenn, C. L. Goodale, L. H. Geiser, C. T. Driscoll, E. B. Allen, J. S. Baron, R. Bobbink, W. D. Bowman, C. M. Clark, B. Emmett, F. S. Gilliam, T. L. Greaver, S. J. Hall, E. A. Lilleskov, L. Liu, J. A. Lynch, K. J. Nadelhoffer, S. S. Perakis, M. J. Robin-Abbott, J. L. Stoddard, K. C. Weathers, and R. L. Dennis. 2011. Effects of nitrogen deposition and empirical nitrogen critical loads for ecoregions of the United States. Ecological Applications 21:3049–3082.

Payne, R. J., N. B. Dise, C. D. Field, A. J. Dore, S. J. M. Caporn, and C. J. Stevens. 2017. Nitrogen deposition and plant biodiversity: past, present, and future. Frontiers in Ecology and the Environment 15:431–436.

Pearse, W. D., M. W. Cadotte, J. Cavender-Bares, A. R. Ives, C. Tucker, S. C. Walker, and M. R. Helmus. 2015. pez: phylogenetics for the environmental sciences. Bioinformatics 31:2888–2890.

R Core Team. 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rao, L. E., and E. B. Allen. 2010. Combined effects of precipitation and nitrogen deposition on native and invasive winter annual production in California deserts. Oecologia 162:1035–1046.

Reich, P. B. 2014. The world-wide "fast-slow" plant economics spectrum: a traits manifesto. Journal of Ecology 102:275–301.

Rosseel, Y. 2012. lavaan: An R package for structural equation modeling. Journal of Statistical Software 48:1–36.

Ryan, J. 2011. Anthoethnography: emerging research into the culture of flora, aesthetic experience of plants, and the wildflower tourism of the future. New Scholar 1:28–40.

Sala, O. E., F. S. III Chapin, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, M. Oesterheld, N. L. Poff, M. T. Skykes, B. H. Walker, M. Walker, and D.

H. Wall. 2000. Global biodiversity scenarios for the year 2100. Science 287:1770–1774.

Samani, Z. A., and M. Pessarakli. 1986. Estimating potential crop evapotranspiration with minimum data in Arizona. Transactions of the ASAE 29:522–524.

Schade, J. D., and S. E. Hobbie. 2005. Spatial and temporal variation in islands of fertility in the Sonoran Desert. Biogeochemistry 73:541–553.

Schlesinger, W. H., and A. M. Pilmanis. 1998. Plant-soil interactions in deserts. Biogeochemistry 42:169–187.

Schlesinger, W. H., J. A. Raikes, A. E. Hartley, and A. F. Cross. 1996. On the spatial pattern of soil nutrients in desert ecosystems. Ecology 77:364–374.

Seto, K. C., M. Fragkias, B. Güneralp, and M. K. Reilly. 2011. A meta-analysis of global urban land expansion. PLoS ONE 6: e23777.

Smith, S. A., and J. W. Brown. 2018. Constructing a broadly inclusive seed plant phylogeny. American Journal of Botany 105:302–314.

Snyman, H. A. 2002. Short-term response of rangeland botanical composition and productivity to fertilization (N and P) in a semi-arid climate of South Africa. Journal of Arid Environments 50:167–183.

Soga, M., and K. J. Gaston. 2016. Extinction of experience: The loss of human-nature interactions. Frontiers in Ecology and the Environment 14:94–101.

Sponseller, R. A., S. J. Hall, D. P. Huber, N. B. Grimm, J. P. Kaye, C. M. Clark, and S. L. Collins. 2012. Variation in monsoon precipitation drives spatial and temporal patterns of *Larrea tridentata* growth in the Sonoran Desert. Functional Ecology 26:750–758.

Taasoobshirazi, G., and S. Wang. 2016. The performance of the SRMR, RMSEA, CFI, AND TLI: an examination of sample size, path size, and degrees of freedom. Journal of Applied Quantitative Methods 11:31–41.

The Plant List. 2013. Version 1.1. http://www.theplantlist.org/.

Threlfall, C. G., and D. Kendal. 2018. The distinct ecological and social roles that wild spaces play in urban ecosystems. Urban Forestry & Urban Greening 29:348–356.

Tucker, C. M., M. W. Cadotte, S. B. Carvalho, T. J. Davies, S. Ferrier, S. A. Fritz, R. Grenyer, M. R. Helmus, L. S. Jin, A. O. Mooers, S. Pavoine, O. Purschke, D. W. Redding, D. F. Rosauer, M. Winter, and F. Mazel. 2017. A guide to phylogenetic metrics for conservation, community ecology and macroecology. Biological Reviews 92:698–715.

UNEP. 1992. World Atlas of Desertification. Edward Arnold, New York, New York, USA.

United Nations. 2014. World Urbanization Prospects: The 2014 Revision.

USDA NRCS. 2020. The PLANTS database. National Plant Data Team, Greensboro, North Carolina, USA. http://plants.usda.gov.

Venable, D. L. 2007. Bet hedging in a guild of desert annuals. Ecology 88:1086–1090.

Venable, D. L., and C. E. Pake. 1999. Population ecology of Sonoran Desert annual plants. Pages 115–142 *in* R. H. Robichaux, editor. Ecology of Sonoran Desert Plants and Plant Communities. University of Arizona Press.

Venable, D. L., C. E. Pake, and A. C. Caprio. 1993. Diversity and coexistence of Sonoran Desert winter annuals. Plant Species Biology 8:207–216.

Webb, C. O., D. D. Ackerly, M. A. McPeek, and M. J. Donoghue. 2002. Phylogenies and community ecology. Annual Review of Ecology and Systematics 33:475–505.

Yahdjian, L., L. Gherardi, and O. E. Sala. 2011. Nitrogen limitation in arid-subhumid ecosystems: a meta-analysis of fertilization studies. Journal of Arid Environments 75:675–680.

Yu, G., D. Smith, H. Zhu, Y. Guan, and T. T.-Y. Lam. 2017. ggtree: an R package for visualization and annotation of phylogenetic trees with their covariates and other associated data. Methods in Ecology and Evolution 8:28–36.

Zanne, A. E., D. C. Tank, W. K. Cornwell, J. M. Eastman, S. A. Smith, R. G. FitzJohn, D. J. McGlinn, B. C. O'Meara, A. T. Moles, P. B. Reich, D. L. Royer, D. E. Soltis, P. F. Stevens, M. Westoby, I. J. Wright, L. Aarssen, R. I. Bertin, A. Calaminus, R. Govaerts, F. Hemmings, M. R. Leishman, J. Oleksyn, P. S. Soltis, N. G. Swenson, L. Warman, and J. M. Beaulieu. 2014. Three keys to the radiation of angiosperms into freezing environments. Nature 506:89–92.

TABLE 1. Experimental sites.

Region	Site Code[1]	Site Name	Site Elevation (m)	Average Precipitation (mm)[2]
Urban	DBG	Desert Botanical Garden	396	78
Urban	MVP	Mountain View Park (North Mountain)	397	71
Urban	PWP	Piestewa Peak (Phoenix Mountain Preserve)	456	76
Urban	SME	South Mountain Park, East	372	62
Urban	SMW	South Mountain Park, West	458	59
Non-urban (East)	LDP	Lost Dutchman State Park	620	132
Non-urban (East)	MCN	McDowell Mountain Regional Park, North	476	115
Non-urban (East)	MCS	McDowell Mountain Regional Park, South	539	102
Non-urban (East)	SRR	Salt River Recreation Area	434	120
Non-urban (East)	UMP	Usery Mountain Regional Park	592	95
Non-urban (West)	EME	Estrella Mountain Regional Park, East	331	55
Non-urban (West)	EMW	Estrella Mountain Regional Park, West	382	53
Non-urban (West)	SNE	Sonoran Desert National Monument, East	492	52
Non-urban (West)	SNW	Sonoran Desert National Monument, West	375	55
Non-urban (West)	WTM	White Tank Mountain Regional Park	454	73

[1] For additional site characteristics, see Hall et al. (2011) and Cook et al. (2018).

[2] Mean winter growing season precipitation (October - March) for the period 2006 to 2018

TABLE 2. Results of model averaging for the top eight generalized linear mixed models predicting annual plant species richness, with site as a random factor.

Predictor	Estimate	Standard error	z value	p value	# models with variable
Current season water availability	-1.3	0.3	4.00	0.0001	8
Microhabitat	0.05	0.03	1.58	0.12	8
N	-0.26	0.03	8.74	<0.0001	8
Previous season water availability	-5.7	0.5	11.18	<0.0001	8
Urban	-1.4	0.3	5.42	<0.0001	8
Current season water availability × Previous season water availability	10	1	8.40	<0.0001	8
Current season water availability × Urban	1.8	0.3	5.21	<0.0001	8
Previous season water availability × Urban	2.9	0.7	4.42	<0.0001	8
P	-0.01	0.03	0.36	0.72	3
P × Urban	-0.01	0.04	0.33	0.74	1
Current season water availability × Microhabitat	0.02	0.07	0.24	0.81	1
Microhabitat × Urban	-0.01	0.02	0.22	0.83	1
Current season water availability × N	0.01	0.05	0.16	0.87	1
Urban × N	0.00	0.02	0.16	0.87	1
Current season water availability × P	0.01	0.06	0.22	0.83	1

Notes: The global model included all two-way interaction terms, but variables not included in any of the eight best models (see methods) are not shown in this table.

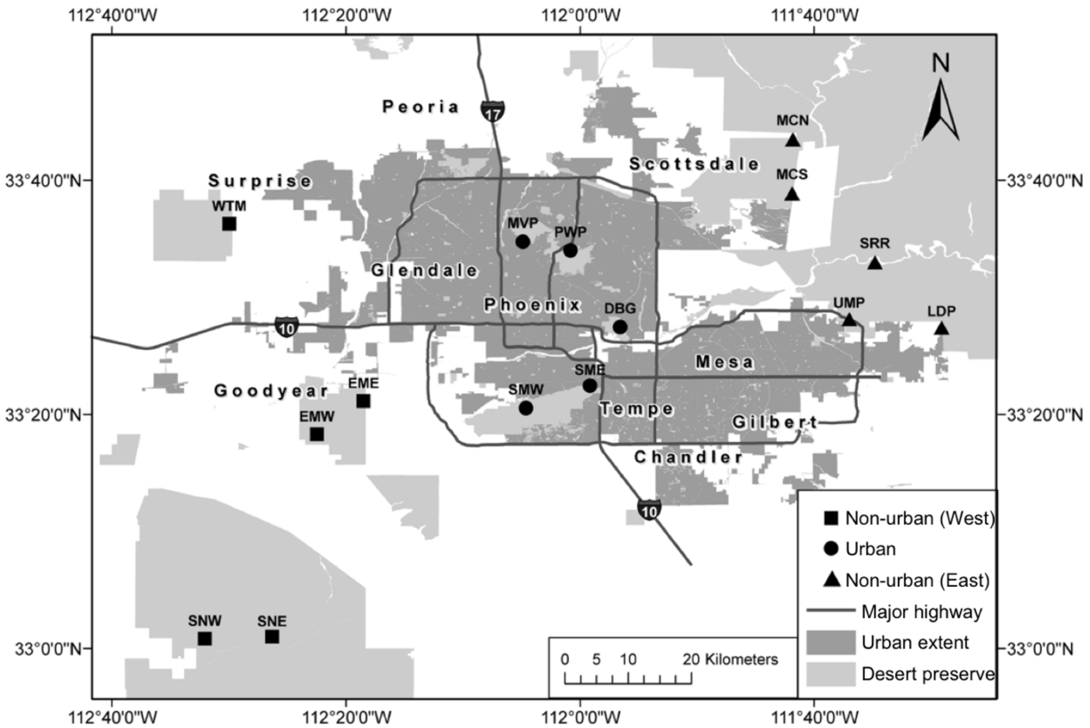

FIG. 1. Map of study sites in and around the Phoenix metropolitan area, adapted from Hall et al. (2011). All sites are located in Sonoran Desert preserves.

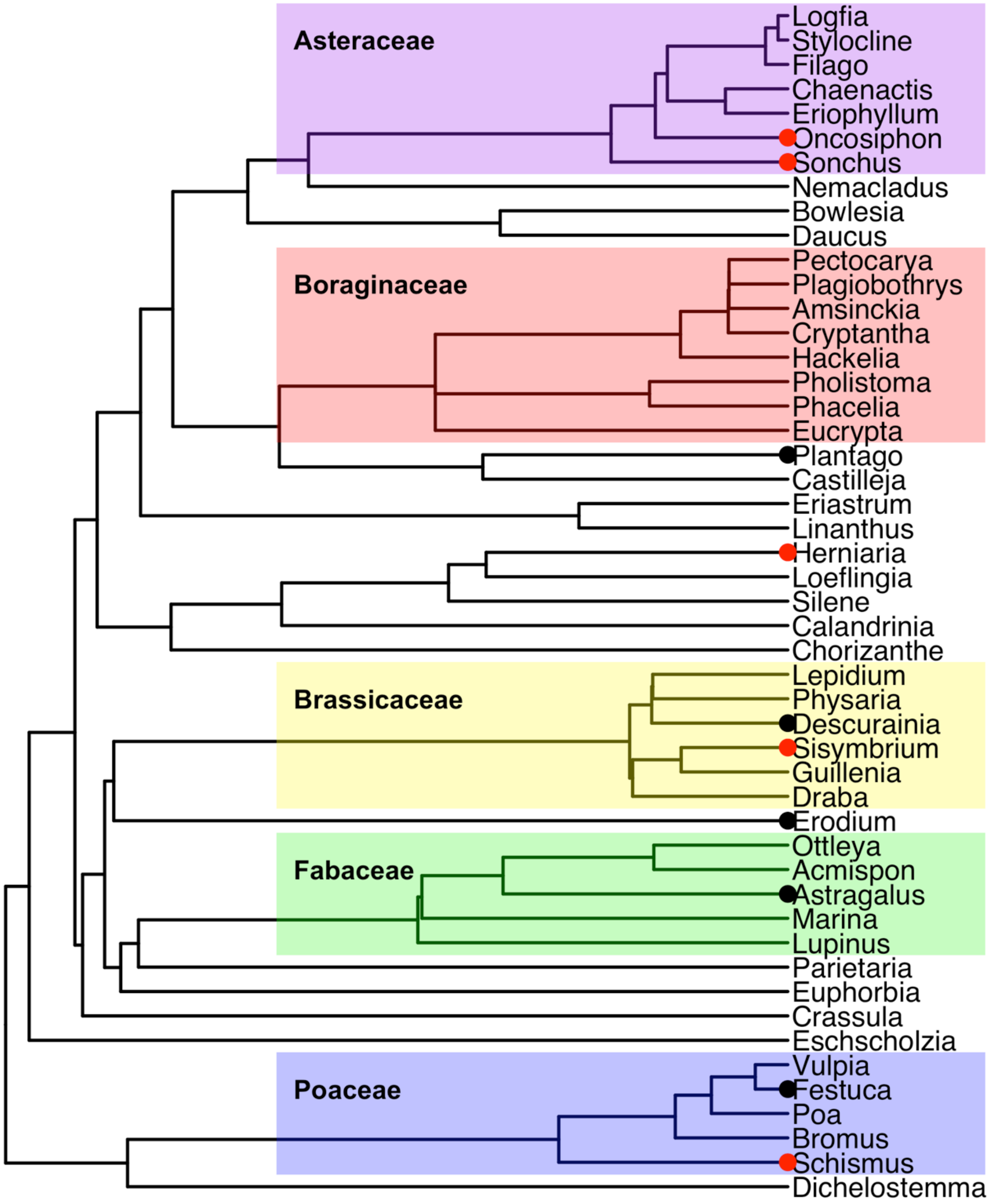

FIG. 2. Phylogenetic tree including all genera observed in at least 0.5% of all samples in this study (see Appendix B: Fig. S1 for complete tree). Tree is adapted from the ALLMB tree defined by Smith and Brown (2018). Colors represent family groupings for families with at least five genera observed. Red points show genera with only non-native species observed and black points show genera with some native and some non-native species. Genera with no points include only native species in this community.

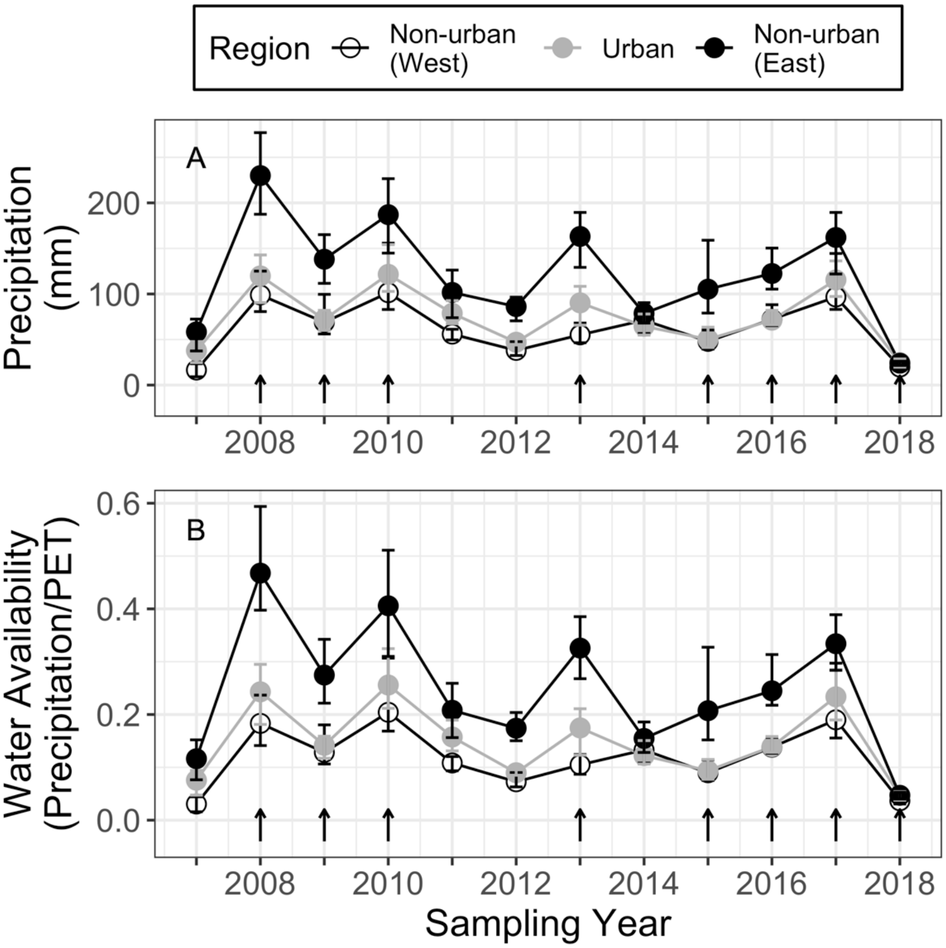

FIG. 3. Precipitation (A) and water availability (B) recorded during the winter annual plant growing season (October - March). Points show the mean value for the five experimental sites within each region, and error bars show the range for a given region and year. Arrows above the x-axis indicate years in which the annual plant community was sampled.

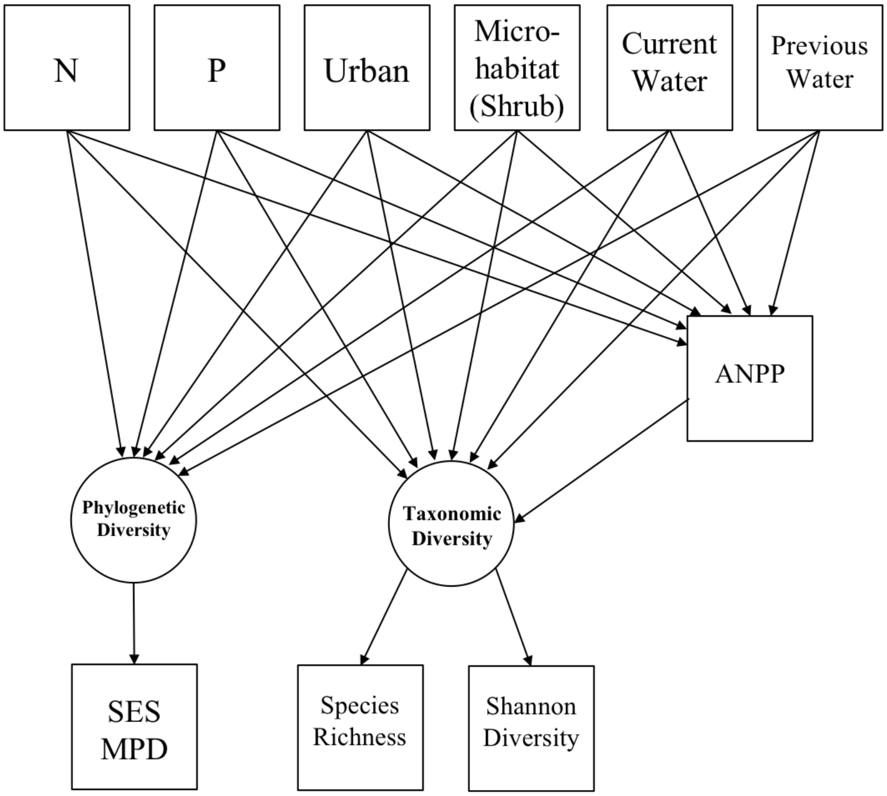

FIG. 4. Theoretical path diagram of annual plant diversity as shaped directly by nutrient availability, water availability, microhabitat, and urban or non-urban location, and indirectly by these predictors via change in ANPP. The latent variable Phylogenetic Diversity is defined entirely by the measured variable standardized effect size of mean phylogenetic distance (SES MPD). The loading for species richness onto the latent variable Taxonomic Diversity was fixed to one. N = nitrogen addition, P = phosphorus addition, Urban = sample in an urban site, Microhabitat (Shrub) = sample located under a shrub, Current Water = current growing season water availability, Previous Water = previous growing season water availability, ANPP = aboveground net primary productivity, SES MPD = standardized effect size of mean pairwise distance.

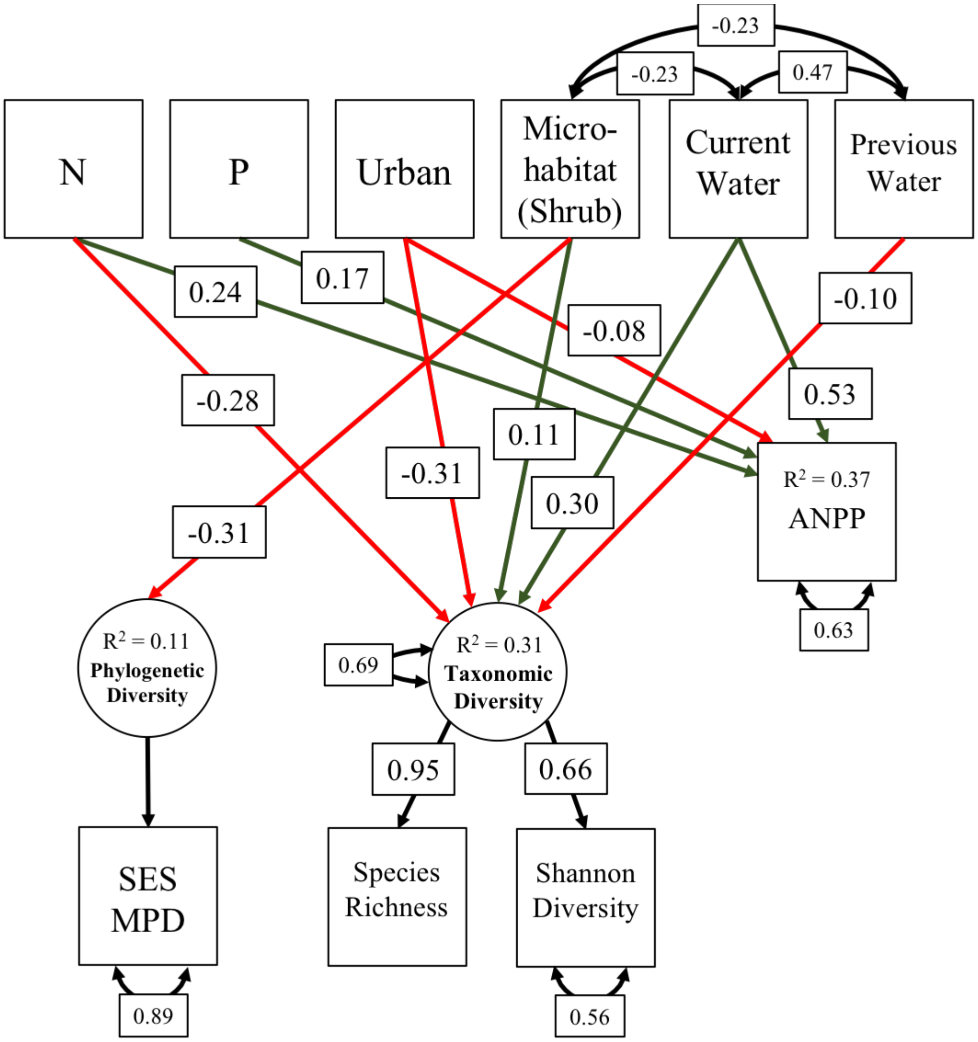

FIG. 5. Structural equation model showing impacts of nutrient enrichment, urban location, microhabitat, and water availability on taxonomic and phylogenetic diversity of annual plants. Green paths indicate positive relationships; red lines indicate negative relationships. Only statistically significant ($p < 0.05$) paths are shown, and values in boxes on paths give standardized model regression coefficients (see Appendix B: Table S5 for non-standardized coefficients). The theorized path between productivity and taxonomic diversity was not statistically significant and so does not appear in the diagram. Double-headed arrows on individual boxes show residual variation in response variables. Double-headed arrows between boxes show correlation between predictors. N = nitrogen addition, P = phosphorus addition, Urban = urban location, Microhabitat (Shrub) = sample located under a shrub, Current Water = current growing season water availability, Previous Water = previous growing season water availability, ANPP = aboveground net primary productivity, SES MPD = standardized effect size of mean pairwise distance.

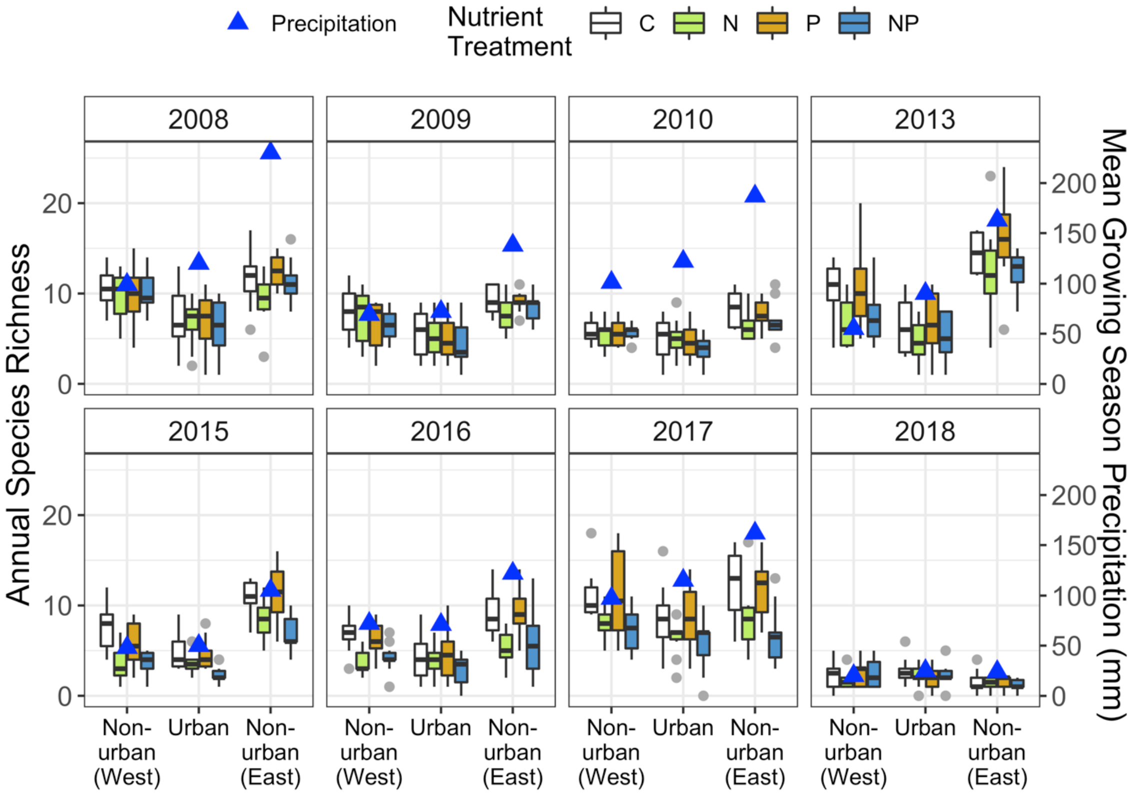

FIG. 6. Annual plant species richness by nutrient enrichment treatment, region, and year. Boxes show species richness with grey points indicating outliers. Blue triangles show the average growing season precipitation for a given region and year. C = control/no nutrient addition, N = nitrogen addition, P = phosphorus addition, NP = nitrogen and phosphorus addition.

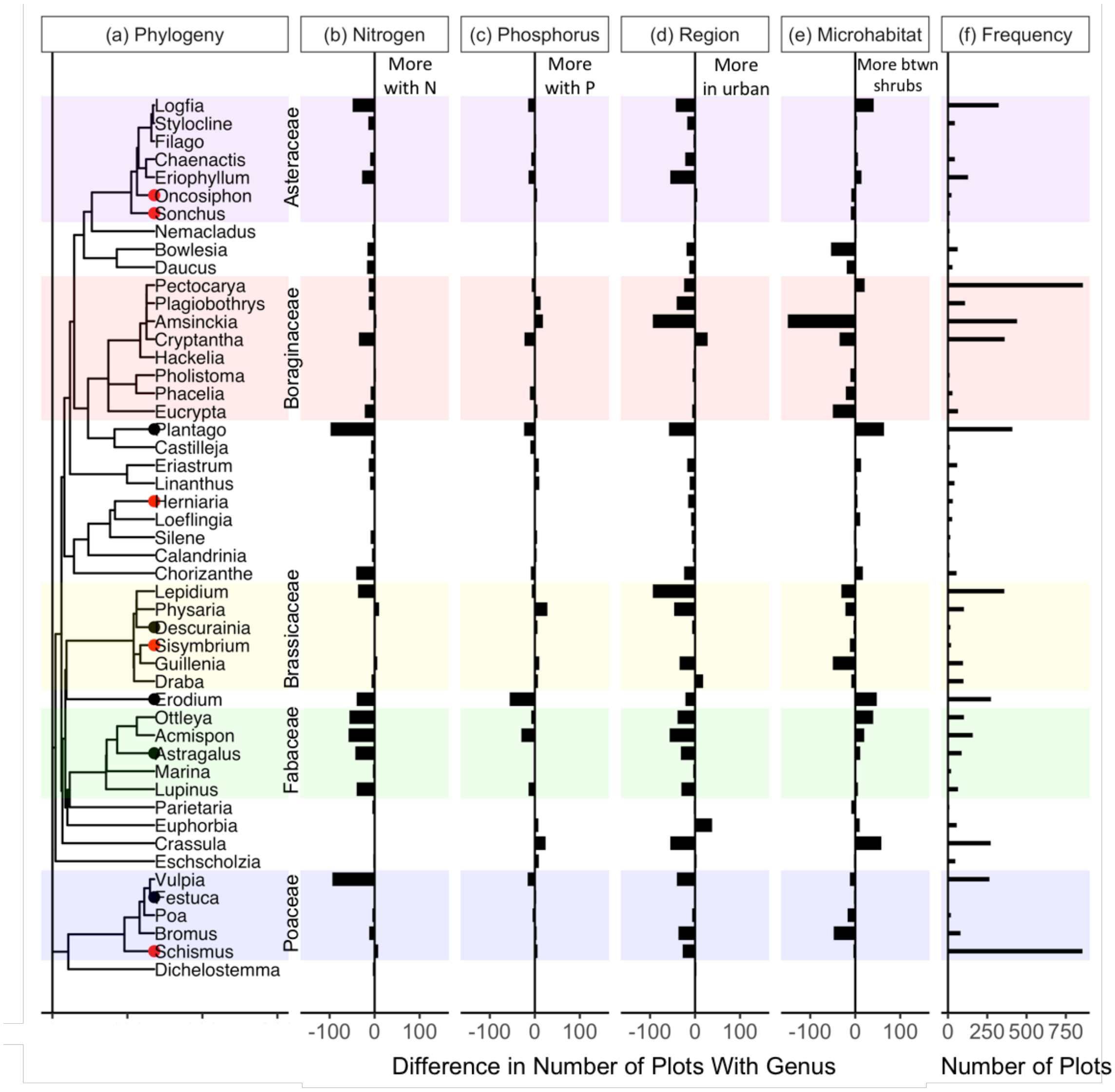

FIG. 7. Difference in number of plots containing each genus by (b, c) N or P enrichment, (d) urban or non-urban location, and (e) microhabitat (under or between shrubs), out of 952 total plots. Note that there were twice as many non-urban as urban samples, so the number of non-urban plots containing each genus was divided by two. Panel (f) shows the total number of plots containing each genus. Plot background shading shows family groupings. Dots on genus names indicate that the genus contained all non-native species (red dot) or some non-native species (black dot).

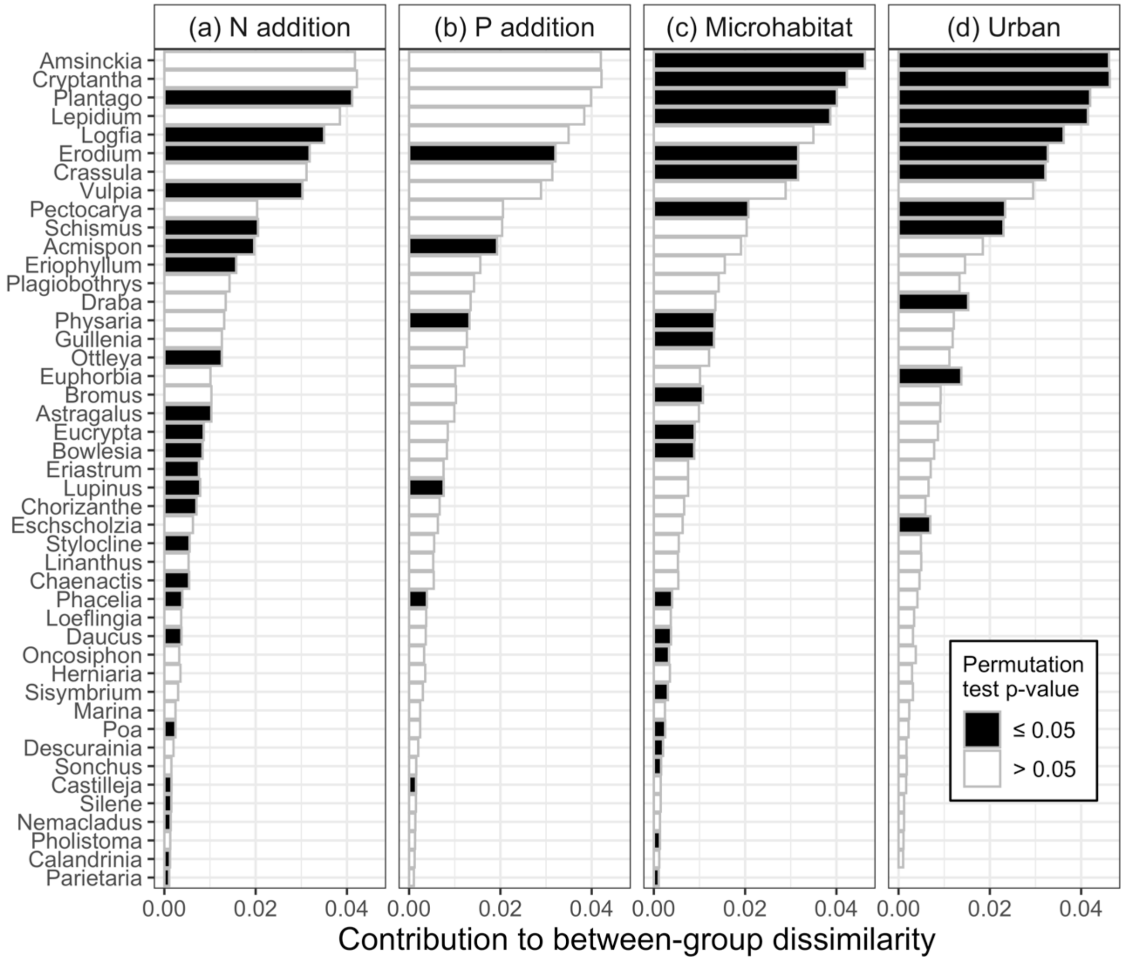

FIG. 8. Contribution of genera to community dissimilarity between plots (a, b) with and without N or P enrichment, (c) under or between shrubs, and (d) in urban or non-urban locations (from SIMPER analysis). Black bars show species that contributed significantly to community dissimilarity at the p ≤ 0.05 level while hollow bars were not statistically significant.

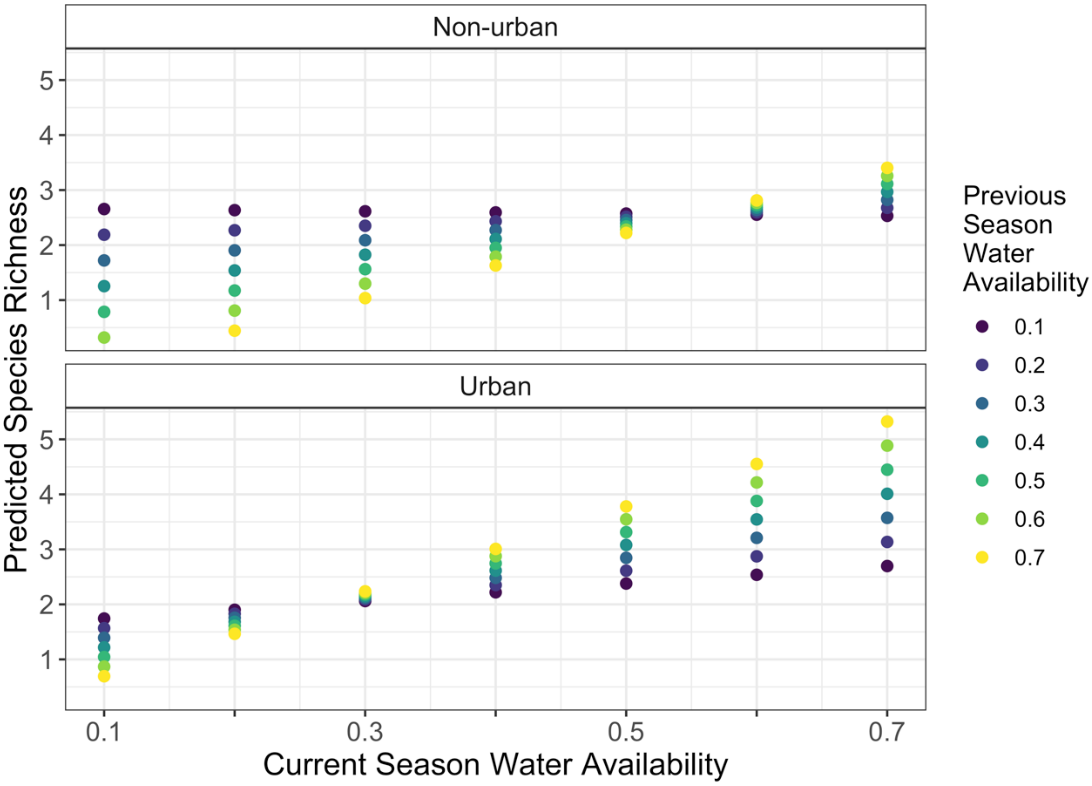

FIG. 9. Model-predicted annual plant species richness showing interactions between current season water availability, previous season water availability, and urban location. Predicted values are shown only to illustrate the modeled interaction terms and are not forecasts. Predictions were generated using the top generalized linear mixed model with a hypothetical dataset containing pairwise combinations of previous and current season water availability across the observed range in urban and non-urban locations. All other variables were held constant, with no N and P addition, under-shrub microhabitat, and random axis for site UMP.

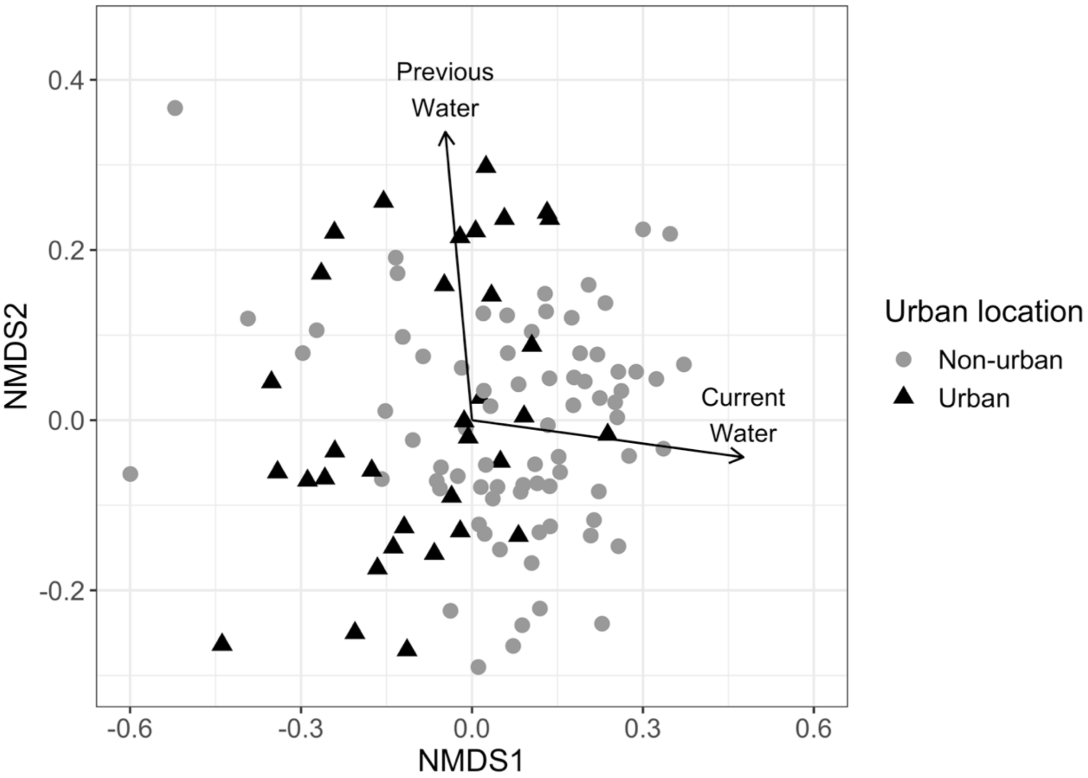

FIG. 10. NMDS of control plots only, based on presence/absence of genera. Composition of plots in different microhabitats is combined. NMDS stress = 0.16. Environmental vectors are significant as determined by permutation tests, with current water $R^2 = 0.23$, $p = 0.001$ and previous water $R^2 = 0.12$, $p = 0.001$.

CHAPTER 4

URBAN RESIDENT ACTIONS INFLUENCE DYNAMIC PLANT COMMUNITIES AND CONSERVATION OPPORTUNITIES

Abstract. Integrated social and ecological processes shape urban plant communities, but the temporal dynamics and potential for change in these managed communities have rarely been explored. In residential yards, which cover about 40% of urban land area, individuals make decisions that control vegetation outcomes. These decisions may lead to static plant composition and structure, as residents seek to expend little effort and maintain consistent, neat landscapes. Alternatively, residents may actively change plant communities to meet their preferences or address yard problems. In this research, we ask, how and to what extent does managed residential yard vegetation change over time? We conducted co-located ecological surveys of yards (in 2008, 2018, and 2019) and social surveys of residents (in 2018) in four diverse Phoenix, Arizona, neighborhoods. 94% of residents had made some changes to their front or back yards since moving in. On average, about 60% of woody vegetation per yard changed between 2008 and 2018, though the number of species present did not change significantly. In comparison, about 30% of woody vegetation change in reference native Sonoran Desert areas over ten years. In yards, change in a single year was much lower than over the ten-year period, with about 15% woody vegetation change on average but up to 90% change in some yards. These results indicate that residential yard plant communities are dynamic and experience a combination of incremental changes and less frequent but substantial changes, such as when homes are sold. We observed greater vegetation change in the two older, lawn-dominated neighborhoods surveyed, despite differences in neighborhood socioeconomic

factors, suggesting that neighborhood age and other characteristics may be important drivers of change while socioeconomic status neither promotes nor inhibits change at the neighborhood scale. Our findings highlight an opportunity for management interventions, wherein residents may be open to making conservation-friendly changes if they are already altering the composition of their yards.

INTRODUCTION

Cities globally are rapidly expanding, with most urban land under human management (Foley et al. 2005, Seto et al. 2011). Management choices in public and private spaces can improve biodiversity outcomes and human wellbeing (Dearborn and Kark 2010, Hartig et al. 2014). In particular, management of urban plant communities can increase the provision of aesthetic and cultural services, wildlife habitat, and cooling services (Jenerette et al. 2011, Robinson and Lundholm 2012, Nesbitt et al. 2017). As interest in these services grows, it is essential to understand how and why urban land managers choose to incorporate new plant communities or change existing ones.

Urban plant communities are often perceived to be relatively static, with evidence of structural and ecological legacies shaping urban form and function over centuries (e.g., Grove et al. 2017, Roman et al. 2018). In contrast, plant communities in natural areas have been shown to be dynamic, changing randomly or directionally over time through processes of disturbance and succession, as well as in response to trophic interactions and other environmental drivers (Clements 1916, Pickett et al. 2009, Pulsford et al. 2016). Urbanization changes vegetation communities, including through slow, continued species loss over time even after the initial effects of conversion (Nowak and Walton 2005, Rogers et al. 2009, Walker et al. 2009, Dolan et al. 2011, Ziter et al. 2017). Changes among urban land use types also alter vegetation, such as with modifications to urban greenspace size or shape over time (reviewed in Gaston et al. 2013) or land abandonment and revitalization (Pearsall and Christman 2012, Johnson et al. 2018). The temporal dynamics of urban tree communities have been well studied in comparison to other types of vegetation, with declining canopy cover over time recorded in many cities (Nowak and Greenfield 2012, 2018, Roman et al. 2017, 2018, Guo et al. 2018, Smith et al. 2019). However, the extent to which entire urban plant communities experience changes in composition over time outside of land-use conversion effects has rarely been evaluated. A better understanding of the temporal dynamics of urban vegetation is important for conservation interventions and realistic community planning.

Residential yards are a model system for understanding how numerous land-management decisions by individuals lead to local- and city-scale vegetation patterns over time. Residential yards compose 25-40% of land area in cities (Loram et al. 2007, Mathieu et al. 2007), and in the U.S., they represent a homogeneous, continental-scale macrosystem with relatively similar biophysical properties compared to natural ecosystems (Groffman et al. 2017). However, individual management decisions made by residents (i.e., homeowners or renters who reside in a home) can influence local-scale ecological outcomes, such as resources available to support wildlife. Decisions range from small, daily choices, like whether and how much to irrigate or remove weeds, to larger, infrequent choices, like whether to build a new pool, remove a lawn, or hire a landscaper. Numerous studies have addressed the variety of attitudinal and social-economic forces driving land manager decisions in urban, residential, and other private land-management contexts (Larson et al. 2010, Drescher et al. 2017). However, the ways in which these decisions result in change – or lack of change – in residential urban vegetation remain unclear.

Following what we know about how individuals make management choices, there is reason to expect maintenance of a static environment in residential yards over time. Residents manage yards to maintain a neat, aesthetically pleasing appearance (Larson et al. 2009, Nassauer et al. 2009, Larson and Brumand 2014), which requires regular maintenance such as pruning and replacement of dead plants. Additionally, residents tend to prioritize low-maintenance yard landscaping, suggesting that they may choose to expend minimal effort by maintaining existing vegetation rather than making changes (Martin et al. 2003, Larson et al. 2009, Conway 2016). Thus, management efforts may result in both structural consistency of the yard over time and a consistent community composition if residents replace removed plants with the same species. Normative pressures and formal regulations to match the neighborhood may also prevent major vegetation change and require upkeep of consistent landscaping (Nassauer et al. 2009, Blaine et al. 2012, Hunter and Brown 2012). These social pressures are particularly influential in the visible front yard, where neighbors are perceived as more likely to view and critique yard management choices (Zmyslony and Gagnon 1998, Hunter and Brown 2012). For example, the normatively prescribed residential lawn requires management to maintain a consistent height, composition, and color (Robbins 2007, Burr et al. 2018), reducing variability or change. Finally, while residents may wish to alter their landscapes, their actual and perceived ability to make desired changes may be low (Goddard et al. 2013, Conway and Vander Vecht 2015, Martini and Nelson 2015, Avolio et al. 2018).

Renters, in particular, may have little control over their outdoor landscaping. Decisions of previous residents, original developer choices, and historical regulatory structures continue to influence present-day landscapes through so-called legacy effects, suggesting that change over time in residential landscapes is slow and relatively minor (Clarke et al. 2013, Grove et al. 2017, Larson et al. 2017b, Roman et al. 2018). Thus, managed plant communities in residential landscapes may be static over time, with management to maintain consistency in the event of disturbance.

In contrast to literature supporting the idea of relatively static vegetation in residential yards, there is also evidence that residents change urban plant communities. For example, yards in the Phoenix metropolitan area have changed from lawn-dominated landscape types to low water-use landscapes over the past several decades (Frost 2016). The extent to which these changes are driven by residents, housing developers, or others is unclear, as the city has continued to grow. This change has been gradual, as opposed to a similar transformation in California, in which lawns were removed in response to drought and short-term water restrictions (Pincetl et al. 2019). Additionally, plant communities in residential spaces can change in response to economic disturbances resulting in home foreclosures and reduced management (Ripplinger et al. 2017). Thus, it is clear that social and ecological disturbances are able to provoke change in urban vegetation, as they do in natural landscape (Collins et al. 2011). Further evidence of change over time can be found in the tendency for residential yards to match resident preferences (Larsen and Harlan 2006, Wheeler et al. 2020). The match between preferences and realized yards tends to be greater with longer duration of residence in the home, suggesting temporal change (Kendal et al. 2012). At broader scales, remote monitoring has shown fluctuations in urban and exurban tree and lawn cover over time, though the drivers of these dynamics are not clear (Nowak and Greenfield 2012, 2018, Huang et al. 2014). Together, these findings suggest that residents can indirectly alter yard vegetation through changes in management intensity as well as actively changing yards to meet their desires and respond to disturbances like drought, changing norms, or new regulations.

Evaluating the ways in which managed urban vegetation is either static or dynamic requires a social-ecological approach. Land managers are guided by complex combinations of personal, social, and structural pressures (Nassauer et al. 2009, Larson et al. 2010, Cook et al. 2012), and often determine how plant communities respond to disturbances or novel conditions (Nowak 2012). To structure our exploration of how managed urban plant communities change over time, we consider two types of change and disturbance: long-term, chronic change ("presses"), and short-term, discrete change ("pulses"), which can include both ecological and social elements (Fig. 1; Collins et al. 2011). Examples of press disturbances within a residential yard context could include shifts in dominant norms and landscape fashion, climate change, or household demographic change. Pulse disturbances could include change in resident through home sale or rental turnover, damage from storms or other discrete events, rapid change in household economic condition or leisure time through changes in employment, and change in incentive or regulatory structures around residential yard management. All of these disturbances could affect yard plant community composition, either by testing the stability of the landscape if maintained for consistency, or by provoking small management events (e.g., planting or removing a single plant) or large ones (e.g., relandscaping an entire yard). While ecological disturbance events may act directly on the plant community (e.g., trees blown down during a storm or killed by pests), social disturbances act indirectly through either pulse or press changes in management behavior (Fig. 1).

In this paper, we ask, how and to what extent does managed residential yard vegetation change over time? We consider three competing hypotheses for change dominated by either "pulse" or "press" patterns (Fig. 1).

H1: Yards are managed in ways that promote spatial and temporal consistency;

H2: Yard dynamics are dominated by ongoing small-scale changes which accumulate over time, suggesting a greater influence of "press"-type disturbances; or

H3: Yards are mostly managed for homogeneity, but with occasional large "pulse"-type changes.

If yards are primarily managed for consistency (H1), we expect low to no vegetation turnover over time. Alternatively, if press changes dominate in residential landscapes (H2), we expect a higher rate of vegetation turnover over longer time periods and little change in short time periods. Finally, if pulse changes are common (H3), we expect yard vegetation turnover to be bimodally distributed, whereby some yards experience little change and some experience high change in response to discrete disturbance events. To address these hypotheses, we conducted a paired social and ecological survey of residential yards in Phoenix, Arizona. We evaluated front yard woody plant community turnover between 2008, 2018, and 2019 to characterize patterns of change in a managed social-ecological system. We then compared turnover in residential yards to turnover in surrounding desert communities in order to contextualize our findings within the local natural landscape.

METHODS

Study area and sample selection

Phoenix, Arizona is located in the Sonoran Desert in the southwestern United States. Annual rainfall is low in this semi-arid region and summer temperatures are high, with an annual average of 20 cm of precipitation and 109 days over 100° F (38° C) (NOAA 2019a, 2019b). The Phoenix–Mesa–Scottsdale metropolitan area is currently among the fastest growing cities in the U.S. (U.S. Census Bureau 2019a). With rapid population growth has come rapid development, previously on agricultural land but increasingly on newly converted desert. Phoenix has historically been considered an oasis in the desert, with abundant grassy landscaping despite its arid context (Hirt et al. 2008). However, alternative desert-like xeric yards are increasingly common, especially in new developments (Larsen and Harlan 2006, Walker et al. 2009, Frost 2016). Due to the changing characteristics of new construction and the rapid influx of people, the city of Phoenix may be more likely to experience landscape change over time than other cities.

We tested our hypotheses about residential vegetation change within four neighborhoods located along a north-to-south transect in the city of Phoenix (see Larson et al. 2010, Larson and Brumand 2014). Neighborhoods were originally selected as part of the Central Arizona-Phoenix Long-Term Ecological Research (CAP LTER) Phoenix Area Social Survey project (Larson et al. 2010, 2017a), with boundaries defined by 2000 U.S. Census block groups. The four neighborhoods used in this study were selected to represent different combinations of socioeconomic status and dominant landscape type. Two sampled neighborhoods had primarily grassy, mesic-type landscaping and two were primarily xeric, and in each of these landscape groupings one neighborhood had higher median income and educational attainment and one lower (Table 1). Due to the historic grassy character of Phoenix and the more recent shift to desert-like landscaping, the two mesic neighborhoods are also older than the xeric neighborhoods (sampled homes in mesic neighborhoods 16-97 years old in 2018, in xeric neighborhoods 14-33 years old; Table 1). The neighborhoods also varied in their demographics and ratio of owners to renters (Table 1). Notably for this study, one neighborhood ("Historic Palms District") is a designated historic district, and in the early to mid 1900s was considered the "best and most exclusive residential district in Phoenix... highly restricted... occupied by the more affluent business and professional men" (Nelson et al. 2020). Thus, this neighborhood has a long history of affluence and privilege not shared by the other three neighborhoods. Current historic protections restrict visible architectural change to the homes. Palm trees (*Washingtonia* spp.) are maintained in the strip of lawn between the sidewalk and road in this neighborhood, but other vegetation is not restricted by the historic designation. However, strong normative pressures to maintain historic character (including lawns) can be assumed to be present in this particular neighborhood (Larson and Brumand 2014).

Within each selected neighborhood, we randomly chose approximately 100 parcels to sample using Maricopa County Tax Assessor records. We also obtained the construction year and last sale date for each chosen parcel from tax assessor records. At each parcel, we conducted vegetation surveys and social surveys to assess changes made to the yard.

Vegetation surveys

We sampled front yard vegetation at 428 parcels in 2008 (April-August), and conducted follow-up vegetation surveys at 417 of these parcels in 2018 (May-June) and 100 parcels in 2019 (September), such that 100 of the same parcels were sampled in all three years. Some parcels were not resampled because they could not be definitively relocated or were inaccessible at the time of sampling. The 100 parcels sampled in all three years were those with returned social surveys in 2018 (see below).

Front yards were surveyed from the front sidewalk or from the street where there was no sidewalk. We did not enter the property unless invited by the resident. Vegetation in the strip of ground between the front sidewalk and the street, where present, was not considered as part of the front yard (see Appendix D for additional details). For each yard, we determined the overall yard typology and inventoried yard vegetation. Based on previous research (Larsen and Harlan 2006, Larson et al. 2009), the yard typologies were broad classes defined as: mostly grass with some leafy plants and trees ("mesic"), some grass and some crushed rock with plants and trees ("oasis"), mostly crushed rock with desert-like plants and trees ("xeric"), mostly patio area with plants and trees in pots ("paved"), and mostly bare dirt with little vegetation ("bare"). We collected all 2018 yard data using the mobile app platform Fulcrum with a custom designed app for data collection (Spatial Networks, Inc, St. Petersburg, FL). 2008 and 2019 data were collected with paper datasheets and transcribed.

We identified front yard vegetation to the lowest possible taxon from the front sidewalk using gross morphological features and knowledge of locally available horticultural species. Field team leaders from 2008, 2018, and 2019 compared plant survey methods to ensure consistency in sampling effort. Only trees, shrubs, succulents, vines, large ornamental bunch grasses, and planted herbaceous species were identified in the field. We did not attempt to identify turfgrass species or small forbs. Although we identified many individuals to species in the field, we conducted analyses at the genus level due to the difficulty of consistently distinguishing common cultivars and hybrids. Some individuals were left as unknowns (4-5% of woody individuals per year) if they could not be seen well enough to identify, often for plants that were small, recently sheared, far from the front sidewalk, or lacking visible identifying characteristics. Photos of each unknown taxon in 2018 were checked against the 2008 species list for the yard to confirm that they were not any species present in 2008. In 2019, we re-surveyed yards using the 2018 species list in the field and recording abundance and any changes in species composition. In addition to taxonomic identifications, we classified individuals in the field by growth form (tree, shrub, succulent, vine, herbaceous, grass, or groundcover). Some common species could be pruned as either a shrub or small tree (e.g., *Nerium oleander*, *Olea europea*, *Sophora secundiflora*), and other genera included species of multiple growth forms (e.g., *Caesalpinia* includes species of shrubs and small trees, *Ficus* includes trees and vines). For these taxa, growth forms were recorded as observed (Appendix D: Table S1) and individuals of the same genus but different growth forms were combined for composition and diversity analyses.

Due to our focus on change over a ten-year period and the difference in seasonal timing of 2019 sampling, our analyses focus on woody and succulent taxa only, with herbaceous and grass species omitted as they may be replaced on much shorter time scales (e.g., annual flowers which are often removed before our 2008 and 2018 summer sampling period). We define woody species to include trees, shrubs, succulents, and vines, as well as some perennial subshrubs, such as *Asparagus densiflorus*, *Asclepias subulata, Canna* x *generalis,* and *Strelitzia* sp. (Appendix D: Table S1). Species nomenclature follows The Plant List (2013). An exception was made for the common palo verde tree, some species of which were classified in The Plant List as *Parkinsonia* and some as *Cercidium*. Due to the prevalence of a common horticultural hybrid (*Parkinsonia* x 'Desert Museum'), all palo verde trees were identified as *Parkinsonia* sp.

Social survey

In mid-May of 2018, we contacted 425 of the 428 households for which vegetation was surveyed in 2008 by postcard. The postcard alerted them to the study and informed them that we would mail a printed questionnaire in the near future. The three households that were not contacted had addresses from 2008 that could not be uniquely identified. Three weeks later, we mailed each household a hand-addressed 9" x 12" envelope containing a printed survey booklet, stamped and addressed return envelope, and hand-signed cover letter with English on one side and a Spanish translation on the other side. Letters included information to request a Spanish language survey. Households were sent a follow-up postcard one week after the survey was mailed to thank respondents and remind others to return the survey. As an incentive for participation, 15 $25 Visa gift cards were raffled among respondents, detailed in the cover letter. Survey booklets were marked with an individual identifier to allow responses to be linked with plant data. For surveys mailed back with a return address on the envelope, we checked identifiers to confirm accurate linkages between addresses and identifiers, and found no discrepancies. A total of 105 surveys were completed and returned, yielding a response rate of 25.7% (excluding 16 surveys returned as vacant or undeliverable; Table 2). Survey materials were approved by the ASU Institutional Review Board (Appendix C).

In the survey, we asked residents how long they had lived in the home and the extent to which they had made changes to the front and back yard. Responses were 1) I have made no changes; 2) I have made a few changes; 3) I have made a lot of changes; and 4) I have completely redone my yard. We also asked whether residents had planted or removed any trees. Survey respondents were primarily homeowners, with relatively high income and education compared to census demographics (Table 1; Appendix D: Table S2). Therefore, caution should be taken in broadly generalizing resident-reported change. However, front yard vegetation change was measured in a representative sample of the four focal neighborhoods, and thus these measures include a broad range of resident and parcel characteristics (Table 1).

Analysis

One address with a returned social survey was excluded from the vegetation data due to inability to view the entire front yard. An additional returned survey had no sample ID and so could not be linked with vegetation data. Three yards with returned social surveys could not be resurveyed in 2019. Thus, a total of 100 parcels had the complete dataset of 2008, 2018, and 2019 vegetation data and 2018 social survey responses, 103 parcels had 2008 and 2018 vegetation data with social survey responses, and 416 parcels had vegetation data for 2008 and 2018 (Table 2).

All analyses were conducted in R, version 3.6.1 (R Core Team 2019). To find vegetation community change over time (hereafter "turnover"), we calculated community dissimilarity for each yard against itself from 2008 to 2018 (ten-year dissimilarity) and from 2018 to 2019 (one-year dissimilarity). Dissimilarity was calculated at the genus level for woody and succulent plants only, excluding unidentified individuals, and is a conservative estimate of change given that replacement of an individual with a new individual in the same genus would not be considered a change. In order to assess the effects of changes in composition and abundance compared to changes in composition only, we calculated both Bray-Curtis dissimilarity (based on counts of individuals) and Jaccard dissimilarity (based on presence/absence; Schroeder and Jenkins 2018). We also calculated both Bray-Curtis and Jaccard dissimilarity for trees alone, as we might expect that this component of the community would change most slowly. We compared the distributions of dissimilarity values between time periods with a two-sample Kolmogorov-Smirnov test using the ks.test function. In addition to dissimilarity metrics, we calculated the generic richness of each yard as the number of unique woody genera present. We conducted a Student's t-test using the t.test function to test whether change in generic richness of yards differed significantly from zero over the ten-year and one-year periods. We also compared generic richness by neighborhood and year (2008 and 2018) using a two-way ANOVA with Tukey post-hoc test.

To address our hypotheses about "pulse" versus "press" changes, we considered two pulse-type changes that we expect to result in high plant community turnover in residential yards: change in yard typology, and change in ownership. Yard typology change came from our field survey classifications and largely reflected change in dominant groundcover type (i.e., lawn, crushed rock, bare dirt, or paving). Change in ownership was obtained from tax assessor most recent sale dates obtained at the time of field surveys in 2018. We compared Bray-Curtis dissimilarity for yards that had or had not changed typology or ownership between 2008 and 2018 using ordinary least squares (OLS) regression. We also compared neighborhoods in this model to test whether vegetation turnover differed by neighborhood, and included an interaction between typology change and neighborhood to determine whether differences in neighborhood dominant typology and socioeconomic status altered the effect of typology change (e.g., typology change may more often correspond with large-scale planting in wealthier neighborhoods). Model residuals were normally distributed (Shapiro-Wilk test, $p = 0.3$). We also ran a Tukey post-hoc test for pairwise comparisons of neighborhoods following OLS modeling.

To contextualize our findings, we compared our ten-year Phoenix residential vegetation turnover to turnover in the Phoenix metropolitan area more broadly and in the native Sonoran Desert around the city. To make these comparisons, we used data from the CAP LTER Ecological Survey of Central Arizona (ESCA; Childers et al. 2018). The ESCA, conducted in 2000, 2005, 2010, and 2015, included vegetation surveys in approximately 200 30 m x 30 m sampling plots distributed in a stratified random pattern across metropolitan Phoenix. Sampling sites included in this analysis were classified as either desert or urban land uses, with urban land uses including residential, industrial, commercial, open (i.e., golf courses, parks, and vacant land), and transportation uses (i.e., highways, roads, and airports). In each ESCA survey, all trees and other perennial vegetation within the plot were identified to species and the number of individuals was recorded.

Using the ESCA data, we compared trees, shrubs, succulents, and vines at the genus level between surveys in 2000 and 2010 and between surveys in 2005 and 2015 to calculate ten-year Bray-Curtis dissimilarity, following the same methods as with our residential yard vegetation. We excluded any individuals not identified to genus or lower. For each plot, we considered whether the dominant land use had changed during each ten-year period, and if not, whether it was primarily a desert or urban plot. Some comparisons over years were not possible because sampling was not conducted at a particular site for one of the compared years, or because dominant land use was not recorded. We included a total of 110 plots (39 desert, 48 urban, and 23 changed) for the 2000-2010 comparison and 112 plots (37 desert, 57 urban, and 18 changed) for the 2005-2015 comparison. We compared the dissimilarity distribution of desert ESCA plots with urban ESCA plots and with our residential front yards using a two-sample Kolmogorov-Smirnov test.

RESULTS

Reported yard change

Most surveyed residents reported having made at least a few changes to their front yard (~75% of respondents) or back yard (~90%) since moving in (Fig. 2). 56% of survey respondents had lived in their current home for at least 10 years. Respondents who had lived in the home for less than 5 years had made only slightly fewer changes than the overall sample, with 66% reporting changes to the front yard and 79% to the back yard. Only 6 respondents reported having made no changes to either the front or the back yard. Considering the tree community alone, over 60% of respondents had both planted and removed at least one tree since moving into their home, and only 16% had neither planted nor removed a tree. While reported change from social survey respondents represented primarily homeowners and higher-income residents, a broader range of households was captured by vegetation sampling.

Vegetation change

Yard plant richness did not significantly change from 2008 to 2018 (t-test, p = 0.13, N = 416), with a median of seven woody plant genera per yard in both years. There was also no significant change in richness from 2018 to 2019 (t-test, p = 0.09, N = 100), with a median of eight woody plant genera in both years. While genus richness did not change over time, it was significantly lower in the two lower-income neighborhoods (Old Hispanic Core and New Xeric Tracts) compared to the two higher-income neighborhoods (Fig. 3).

The mean Bray-Curtis dissimilarity of a site with itself between 2008 and 2018 was 0.58 for all woody vegetation (equivalent to 58% turnover; median = 0.60; Fig. 4). Only 3 yards (0.7% of sample) had the same vegetation composition in both years (dissimilarity = 0), while 33 yards (8% of sample) experienced complete turnover (dissimilarity = 1). Measured dissimilarity was somewhat higher where residents reported having made more changes, though this difference was not statistically significant (Appendix D: Fig. S2). Mean Jaccard dissimilarity (presence/absence only) from 2008 to

2018 was 0.62, with 11 yards (3%) with dissimilarity of 0 and 33 yards (8%) with dissimilarity of 1 (Appendix D: Fig. S3). Results for Bray-Curtis and Jaccard dissimilarity show comparable trends (Appendix D: Table S3); therefore, hereafter we primarily interpret results using Bray-Curtis dissimilarity.

Turnover in a single year (2018-2019) was significantly lower than over a ten-year period (Bray-Curtis dissimilarity, mean = 0.18, median = 0.13; Two-sample Kolmogorov-Smirnov test with 100 resampled yards only, D = 0.66, $p < 0.0001$; Fig. 4). However, few yard plant communities remained exactly the same over this year: 7 out of 100 yards had no change in woody plant composition or abundance (Bray-Curtis dissimilarity = 0; Fig. 4) and 31 of 100 yards had no change in genera present (Jaccard dissimilarity = 0; Appendix D: Fig. S3). Meanwhile, 5% of yards experienced over 50% turnover during this year.

Average turnover of trees in a ten-year period was similar to turnover of all woody plants (mean Bray-Curtis dissimilarity = 0.54; Appendix D: Fig. S4). However, tree communities more often experienced either no change or complete change. 16% of yards had entirely the same tree communities in 2008 and 2018, compared to less than 1% with the same complete woody plant community. 27% of yards had entirely different tree communities in 2008 and 2018, compared to 8% of yards with complete turnover of all woody plants. Most yards had very few trees relative to total number of woody plants. In 2018, 15% of yards had no trees at all, 23% had one tree, and 42% had 2-5 trees, with 3 trees per yard on average (median 2, range 0-31). In contrast, yards in 2018 had on average 11 shrubs (median 7, range 0-117) and 5 succulents (median 2, range 0-58).

Influence of "pulse" changes and neighborhood characteristics

Vegetation turnover was higher in yards that experienced "pulse" changes (home sale or change in typology) than in those that did not (Table 3, Fig. 5). Yards with changes in typology experienced significantly higher turnover of woody vegetation compared to yards that did not change typology ($p = 0.002$), and homes that had sold in the period from 2008 to 2018 also experienced greater turnover ($p < 0.0001$).

Turnover differed by neighborhood, with greater vegetation community dissimilarity over time in the two primarily mesic neighborhoods (Old Hispanic Core and Historic Palms District) compared to the primarily xeric neighborhoods (Table 3, Fig. 6). Neighborhoods with the same dominant landscape type but different socioeconomic characteristics did not have significantly different vegetation turnover (Fig. 6). Yard typology change resulted in significantly less vegetation turnover in the Old Hispanic Core (compared to Wealthy Mountain Oasis, significant interaction effect, $p = 0.008$; Table 3), indicating that changes in typology in this neighborhood were not generally coupled with either planting or removal of woody vegetation. Most typology changes in the Old Hispanic Core were conversions from lawn to bare dirt, likely due to reduced maintenance.

Woody plant turnover in residential and desert land uses

Woody vegetation in urban ESCA sites experienced high turnover similar to our observations in residential yards (mean Bray-Curtis dissimilarity = 0.72 for 2000-2010; 0.73 for 2005-2015). In comparison, desert woody plant communities had only 25-30% community turnover in a ten-year period (mean Bray-Curtis dissimilarity = 0.26 for

2000-2010; 0.30 for 2005-2015). Community dissimilarity distributions differed significantly between desert and urban ESCA plots (Two-sample Kolmogorov-Smirnov test; 2000-2010, D = 0.76, $p < 0.0001$; 2005-2010, D = 0.75, $p < 0.0001$; Fig. 7) and between desert ESCA plots and our sampled residential yards (Two-sample Kolmogorov-Smirnov test; 2000-2010 desert and 2008-2018 residential, D = 0.59, $p < 0.0001$; 2005-2015 desert and 2008-2018 residential, D = 0.52, $p < 0.0001$). ESCA plots that experienced landscape type conversion had very high woody plant community turnover (mean Bray-Curtis dissimilarity = 0.92 for 2000-2010; 0.84 for 2005-2015). Most conversions were of agricultural, desert, vacant, or open space into urban or open space (Appendix D: Fig. S5).

DISCUSSION

High temporal turnover of plant communities on privately managed land presents an opportunity for conservation. This study reveals that people regularly make changes to the plant communities they manage, which as a result are dynamic over time. Therefore, there may be opportunities to engage residents in strategies to achieve conservation or ecosystem service benefits (van Heezik et al. 2020). With a better understanding of the motivations for and barriers to specific changes, it may be possible to promote desirable actions to further biodiversity goals in urban areas.

Characterizing yard vegetation turnover

The high turnover we observed in a ten-year period refutes the hypothesis that yard plant communities are static over time. This high turnover is somewhat surprising given previous research showing strong legacy effects on present day vegetation in cities (Pickett et al. 2008, Grove et al. 2017, Larson et al. 2017b, Roman et al. 2018). Legacy effects may originate from slow neighborhood-level change in response to dominant landscape norms, regulatory structures, and socioeconomic stratification (e.g., Grove et al. 2017, Namin et al. 2020), but we find high community turnover at the parcel level, likely in response to household decision-making. Even for tree communities, for which legacy effects are well documented (e.g., Clarke et al. 2013, Conway and Bourne 2013, Roman et al. 2018), we saw high turnover and reported change among residents. It is likely that communities originally planned and developed with more vegetation and greater diversity will persist with these advantages, thus perpetuating differences between historically privileged communities like the Historic Palms District and more marginalized communities like the Old Hispanic Core (Clarke et al. 2013, Grove et al. 2017, Roman et al. 2018, Namin et al. 2020). However, our results show that the vegetation composition can change over time, which presents an opportunity for increased service provision through altered species choices. Further research should investigate how vegetation turnover varies among households with different socioeconomic and attitudinal characteristics to gain greater clarity into the types of changes that are made by which people for what reasons.

We saw evidence of both pulse and press-type changes on vegetation turnover in residential yards over a ten-year and one-year period. The two pulse events we evaluated, yard typology change and home sale during the ten-year period, both resulted in significantly higher vegetation turnover than observed in yards without these pulse events. Additionally, a small number of yards experienced high turnover in a single year, suggesting the occurrence of a pulse of management. Thus, we show that pulse changes can result in high woody vegetation turnover over short and longer time periods. We also expect that ecological pulse disturbances such as severe, localized storms would result in increased turnover more directly, rather than mediated through management actions (Conway and Yip 2016).

Press changes, such as drought, changes in landscape fashions, or accumulating management choices, were also supported by the many yards which experienced small but non-zero turnover in a single year. This finding suggests that ongoing press changes accumulate over time and contribute to high turnover in residential yards. Thus, we conclude that these managed plant communities experience change as both small and large events over time.

Residential vegetation turnover was high, but genus richness of yards did not significantly change over the year or decade. Given our conservative measure of turnover, whereby removal and replacement of a plant with another plant in the same genus would not be considered a change, our turnover results indicate that people remove plants without replacement, add new plants independently, or replace removed plants with others that are not closely related, thus leading to shifts in community composition. The relative stability of genus richness compared to compositional change suggests that replacement of plants with different taxa is most common over time. Identification and analysis of change at the species level may reveal still greater turnover if residents substituted closely related species or cultivars for removed plants, which our genus-level identifications would not have captured. While yards have not become more diverse over time, they also have not lost diversity. Thus, urban areas may remain more diverse than natural ones (Walker et al. 2009) and inequitable distributions of urban biodiversity may persist (Clarke et al. 2013), regardless of high turnover.

Comparison to native desert turnover

We observed higher vegetation turnover in residential yards than in native desert, in part due to resident management. Changing homeownership, which likely corresponds with changes in management, led to greater vegetation turnover, and changing yard typology also provoked plant community change. Shifts in management behavior following a large-scale economic disturbance have been shown to result in changes to the residential plant community in other studies, primarily due to reduced maintenance and growth of weeds (Ripplinger et al. 2016). Additionally, previous research showing that yard plant communities match stated resident preferences suggests that residents make changes to meet their personal ideals (Kendal et al. 2012). We show that managed yard plant communities are dynamic and that even new residents often had made changes.

High turnover in the residential plant community compared to native desert may also be explained by the unique stressors of the urban environment, which could lead to heightened mortality. For example, urban plant communities often experience compacted soil, inappropriate watering or other maintenance practices, and increased prevalence of disease (Scharenbroch et al. 2005, Tubby and Webber 2010, Cook et al. 2012, Roman et al. 2014). Species planted in urban areas may be poorly suited to the local climate and conditions and may be unable to survive or thrive, or planting locations may lead to conflict with infrastructure and lead to removal as plants grow (Martin 2008, Roman et al. 2014, Conway 2016). Higher nighttime temperatures and small planting spaces can also lead to increased mortality rates in urban plant communities (Lu et al. 2010, Vogt et al. 2015). Other work on urban tree communities has documented relatively short lifespans, especially for street trees (Roman and Scatena 2011, Widney et al. 2016, Smith et al. 2019), which may extend to understory woody vegetation as well. Thus, increased mortality rates could explain the high vegetation turnover we observed.

Resident and neighborhood characteristics as drivers of change

The two older, lawn-dominated mesic neighborhoods in our study had higher vegetation turnover than did the two newer, xeric neighborhoods, a pattern that may be driven by either dominant landscape type or neighborhood age. City, neighborhood, and home age have often been identified as important predictors of vegetation diversity, density and cover (Hope et al. 2003, Lowry et al. 2012, Clarke et al. 2013, Aronson et al. 2014). In this study, dominant neighborhood landscape type and neighborhood age covaried, and either could drive the observed higher turnover in the mesic, older neighborhoods. Higher turnover in mesic-style neighborhoods could result from greater planting of species that match the lush character of the landscaping but are unsuitable for the desert climate or from high water provisioning to support lawns that is incompatible with drought-adapted species planted in adjacent garden beds, both leading to high mortality. Alternatively, neighborhood age could drive the observed turnover pattern if replacement of older, outdated landscaping or oversized vegetation leads to greater vegetation change (Kirkpatrick et al. 2013). Additional research to separate the factors driving change would clarify whether these mechanisms each contribute to turnover or whether one is dominant. Understanding the driving mechanisms has implications for management recommendations and ways to harness vegetation change for conservation benefit. For example, if replacement of old vegetation is an important driver, then current fashions and vegetation size may drive the choice of replacement species in any aging neighborhood. Alternatively, if plants in unsuitable local environments have higher mortality rates due to plant selection and management choices, then changes in management practices may decrease turnover, and aging xeric neighborhoods would not be expected to experience increased turnover over time.

Our findings suggest that while wealthier neighborhoods and residents may have more plant diversity, as expected based on the luxury effect, a wide range of neighborhoods experience high vegetation turnover. Socioeconomic status is a well-established predictor of residential plant diversity and abundance (Hope et al. 2003, Grove et al. 2014, Avolio et al. 2015, Gerrish and Watkins 2018, Leong et al. 2018) and is also related to tree survival (Roman et al. 2014, Ko et al. 2015). Neighborhoods with greater financial resources could have less dynamic vegetation, because residents are able to maintain consistent plant communities over time, or conversely, they may have more dynamic vegetation, because residents can afford to make potentially expensive changes like adding and removing trees. We did not see evidence of these differences in turnover, but a combination of these two mechanisms may act at the parcel scale and could obscure one another at the neighborhood level. Additionally, the rate of turnover may be the same among neighborhoods but with different types of change occurring. For example, typology change in the Old Hispanic Core neighborhood was often of lawn to bare dirt as lawns were allowed to die, which could result in death of other nearby vegetation as well. Meanwhile, in the other mesic neighborhood (Historic Palms District), typology conversions were usually of lawn to xeric landscaping, which likely included planting of new desert species. Both instances involve high vegetation turnover, but for different reasons and with different outcomes.

We also expect that turnover differs by who manages a yard. We were not able to determine whether parcels occupied by renters experienced different rates of vegetation turnover compared to those occupied by homeowners due to low response of renters to our resident survey (5% of respondents, compared with 6-48% by neighborhood from Census data; Table 1). However, we expect that renters manage yards with different priorities and have less yard vegetation, which may impact turnover (Landry and Chakraborty 2009, Larson et al. 2009). Parcels managed by landscaping professionals, particularly in higher income areas, may also differ in turnover (Harris et al. 2012). Further research should explore how individual household socioeconomics, homeownership, and delegation of yard management relate to vegetation turnover in order to more fully investigate the reasons and motivations for vegetation change.

In this study, we considered only vegetation in residential front yards. However, we expect that patterns of vegetation turnover may be higher in private back yards. Of our survey respondents, most reported having made changes to the front yard, but an even higher percentage had changed the back yard. Management in front and back yards may differ due to the influence of normative pressures and resident use of public front compared to private back yards (Larsen and Harlan 2006, Nassauer et al. 2009, Larson and Brumand 2014, Locke et al. 2018, Ossola et al. 2019), which could affect vegetation turnover if dying plants are more quickly removed or replaced in the front, or if desired changes can be more freely made in the back. Additionally, front yards may be constrained by formal regulations, especially for yards regulated by homeowners' associations (Martin et al. 2003, Harris et al. 2012, Turner and Stiller 2020). While we did not specifically address the effect of normative constraints or formal regulations in this study, future work comparing front and back yard rates of vegetation change would be well suited to explore the effects of social norms and front yard regulations that may alter vegetation management behavior.

Research implications

We examined vegetation turnover in a single, semi-arid city and over one long and one short time period, but suggest that these results may be more broadly generalizable. Cities in a more mesic environment may see vegetation turnover rates more similar to their respective natural environments, if the stressors of the urban environment or management changes are less extreme. However, a study of tree mortality in Boston rural forests compared to urban street trees found much higher mortality among street trees (Smith et al. 2019) and average street tree lifespan across a range of cities is estimated at 28 years (Roman and Scatena 2011), suggesting that urban vegetation in a variety of climates experience stressors leading to greater mortality rates. Our study was conducted from 2008 to 2018, a time period corresponding with a major economic downturn (2008-2009). Other research in this region showed vegetation responses to this large-scale event, and linked vegetation change to changing economic conditions (Ripplinger et al. 2016, 2017). While we expect that this economic event also influenced our sample sites, we further show non-zero turnover in the year from 2018 to 2019, revealing that vegetation change is not a unique feature of the 2006-2010 period of economic instability. Thus, we suggest that the dynamic community we observed is not unique to this time and place, although further research in other contexts should seek to validate these conclusions.

Further research into the drivers of the high plant turnover observed here will help guide actions to harness vegetation change for positive outcomes. Techniques to reduce turnover caused by mortality from poor management, such as educational campaigns and young tree care assistance, can promote increased canopy cover and ecosystem service provision by larger, older trees (Roman et al. 2015). In conjunction with these efforts, outreach with nurseries, landscaping professionals, and tree planting non-profits can help provide access to native species and species with beneficial characteristics for residents and wildlife (Polakowski et al. 2011, Conway and Vander Vecht 2015). Additionally, resident values and attitudes toward trees and other yard vegetation must be understood and accommodated in any educational or structural change to promote conservation outcomes, with particular attention to historically disadvantaged communities that may both live in neighborhoods with less vegetation and be underrepresented in sampling efforts (Landry and Chakraborty 2009, Yue et al. 2012, Heberlein 2013, Kirkpatrick et al. 2013). External factors such as plant cost and availability, incentivization programs, and local regulations interact with internal drivers including social norms, personal values, and personal capabilities to shape the paths from preferences to outcomes. With a better understanding of these pathways, conservationists can direct resident behaviors towards positive change for biodiversity and human wellbeing outcomes.

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REFERENCES

Aronson, M. F. J., F. A. La Sorte, C. H. Nilon, M. Katti, M. A. Goddard, C. A. Lepczyk, P. S. Warren, N. S. G. Williams, S. Cilliers, B. Clarkson, C. Dobbs, R. Dolan, M. Hedblom, S. Klotz, J. L. Kooijmans, I. MacGregor-Fors, M. McDonnell, U. Mörtberg, P. Pyšek, S. Siebert, J. Sushinsky, P. Werner, and M. Winter. 2014. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. Proceedings of the Royal Society B: Biological Sciences 281:20133330.

Avolio, M. L., D. E. Pataki, S. Pincetl, T. W. Gillespie, G. D. Jenerette, and H. R. McCarthy. 2015. Understanding preferences for tree attributes: the relative effects of socio-economic and local environmental factors. Urban Ecosystems 18:73–86.

Avolio, M. L., D. E. Pataki, T. L. E. Trammell, and J. Endter-Wada. 2018. Biodiverse cities: the nursery industry, homeowners, and neighborhood differences drive urban tree composition. Ecological Monographs 88:259–276.

Blaine, T. W., S. Clayton, P. Robbins, and P. S. Grewal. 2012. Homeowner attitudes and practices towards residential landscape management in Ohio, USA. Environmental Management 50:257–271.

Burr, A., D. M. Hall, and N. Schaeg. 2018. The perfect lawn: exploring neighborhood socio-cultural drivers for insect pollinator habitat. Urban Ecosystems 21:1123–1137.

Childers, D. L., N. B. Grimm, D. Hope, J. P. Kaye, C. A. Martin, N. E. McIntyre, and J. Stutz. 2018. Ecological Survey of Central Arizona: a survey of key ecological indicators in the greater Phoenix metropolitan area and surrounding Sonoran Desert, ongoing since 1999. https://sustainability.asu.edu/caplter/data/view/knb-lter-cap.652/

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

Clements, F. E. 1916. Plant succession: an analysis of the development of vegetation. Carnegie Institution of Washington.

Collins, S. L., S. R. Carpenter, S. M. Swinton, D. E. Orenstein, D. L. Childers, T. L. Gragson, N. B. Grimm, J. M. Grove, S. L. Harlan, J. P. Kaye, A. K. Knapp, G. P. Kofinas, J. J. Magnuson, W. H. McDowell, J. M. Melack, L. A. Ogden, G. P. Robertson, M. D. Smith, A. C. Whitmer. 2011. An integrated conceptual framework for long-term social-ecological research. Frontiers in Ecology and the Environment 9:351–357.

Conway, T. M. 2016. Tending their urban forest: residents' motivations for tree planting and removal. Urban Forestry & Urban Greening 17:23–32.

Conway, T. M., and K. S. Bourne. 2013. A comparison of neighborhood characteristics related to canopy cover, stem density and species richness in an urban forest. Landscape and Urban Planning 113:10–18.

Conway, T. M., and J. Vander Vecht. 2015. Growing a diverse urban forest: species selection decisions by practitioners planting and supplying trees. Landscape and Urban Planning 138:1–10.

Conway, T. M., and V. Yip. 2016. Assessing residents' reactions to urban forest disservices: a case study of a major storm event. Landscape and Urban Planning 153:1–10.

Cook, E. M., S. J. Hall, and K. L. Larson. 2012. Residential landscapes as social-ecological systems: A synthesis of multi-scalar interactions between people and their home environment. Urban Ecosystems 15:19–52.

Dearborn, D. C., and S. Kark. 2010. Motivations for conserving urban biodiversity. Conservation Biology 24:432–440.

Dolan, R. W., M. E. Moore, and J. D. Stephens. 2011. Documenting effects of urbanization on flora using herbarium records. Journal of Ecology 99:1055–1062.

Drescher, M., G. K. Warriner, J. R. Farmer, and B. M. H. Larson. 2017. Private landowners and environmental conservation: a case study of social- psychological determinants of conservation program participation in Ontario. Ecology and Society 22:44.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. III Chapin, M. T. Coe, G. C. Daily, H. K. Gibbs, J. H. Helkowski, T. Holloway, E. A. Howard, C. J. Kucharik, C. Monfreda, J. A. Patz, I. C. Prentice, N. Ramankutty, and P. K. Snyder. 2005. Global consequences of land use. Science 309:570–574.

Frost, D. 2016, January. An Eye on Every Drop. American Planning Association Magazine.

Gaston, K. J., M. L. Ávila-Jiménez, and J. L. Edmondson. 2013. Managing urban ecosystems for goods and services. Journal of Applied Ecology 50:830–840.

Gerrish, E., and S. L. Watkins. 2018. The relationship between urban forests and income: a meta-analysis. Landscape and Urban Planning 170:293–308.

Goddard, M. A., A. J. Dougill, and T. G. Benton. 2013. Why garden for wildlife? Social and ecological drivers, motivations and barriers for biodiversity management in residential landscapes. Ecological Economics 86:258–273.

Groffman, P. M., M. Avolio, J. Cavender-Bares, N. D. Bettez, J. M. Grove, S. J. Hall, S. E. Hobbie, K. L. Larson, S. B. Lerman, D. H. Locke, J. B. Heffernan, J. L. Morse, C. Neill, K. C. Nelson, J. O'Neil-Dunne, D. E. Pataki, C. Polsky, R. Roy Chowdhury, and T. L. E. Trammell. 2017. Ecological homogenization of residential macrosystems. Nature Ecology & Evolution 1:0191.

Grove, J. M., D. H. Locke, and J. P. M. O'Neil-Dunne. 2014. An ecology of prestige in New York City: examining the relationships among population density, socio-economic status, group identity, and residential canopy cover. Environmental Management 54:402–419.

Grove, M., L. Ogden, S. Pickett, C. Boone, G. Buckley, D. H. Locke, C. Lord, and B. Hall. 2017. The legacy effect: understanding how segregation and environmental injustice unfold over time in Baltimore. Annals for the American Association of Geographers 108:524–537.

Guo, T., J. Morgenroth, and T. Conway. 2018. Redeveloping the urban forest: the effect of redevelopment and property-scale variables on tree removal and retention. Urban Forestry & Urban Greening 35:192–201.

Harris, E. M., C. Polsky, K. L. Larson, R. Garvoille, D. G. Martin, J. Brumand, and L. Ogden. 2012. Heterogeneity in residential yard care: evidence from Boston, Miami, and Phoenix. Human Ecology 40:735–749.

Hartig, T., R. Mitchell, S. de Vries, and H. Frumkin. 2014. Nature and Health. Annual Review of Public Health 35:207–228.

Heberlein, T. A. 2013. Navigating environmental attitudes. Conservation Biology 26:583–585.

Hirt, P., A. Gustafson, and K. L. Larson. 2008. The mirage in the valley of the sun. Environmental History 13:482–514.

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

Huang, Q., D. T. Robinson, and D. C. Parker. 2014. Quantifying spatial-temporal change in land-cover and carbon storage among exurban residential parcels. Landscape Ecology 29:275–291.

Hunter, M. C. R., and D. G. Brown. 2012. Spatial contagion: gardening along the street in residential neighborhoods. Landscape and Urban Planning 105:407–416.

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

Johnson, A. L., D. Borowy, and C. M. Swan. 2018. Land use history and seed dispersal drive divergent plant community assembly patterns in urban vacant lots. Journal of Applied Ecology 55:451–460.

Kendal, D., K. J. H. Williams, and N. S. G. Williams. 2012. Plant traits link people's plant preferences to the composition of their gardens. Landscape and Urban Planning 105:34–42.

Kirkpatrick, J. B., A. Davison, and G. D. Daniels. 2013. Sinners, scapegoats or fashion victims? Understanding the deaths of trees in the green city. Geoforum 48:165–176.

Ko, Y., J.-H. Lee, E. G. McPherson, and L. A. Roman. 2015. Factors affecting long-term mortality of residential shade trees: evidence from Sacramento, California. Urban Forestry & Urban Greening 14:500–507.

Landry, S. M., and J. Chakraborty. 2009. Street trees and equity: evaluating the spatial distribution of an urban amenity. Environment and Planning A 41:2651–2670.

Larsen, L., and S. L. Harlan. 2006. Desert dreamscapes: residential landscape preference and behavior. Landscape and Urban Planning 78:85–100.

Larson, K. L., R. Andrade, A. York, D. Childers, P. Coseo, S. Earl, M. Ehlenz, N. Grimm, S. J. Hall, S. Harlan, D. Hondula, S. B. Lerman, D. Pfeiffer, J. Ripplinger, K. Smith, P. Warren, M. Watkins, D. White, S. Wittlinger, M. Wright, and A. Wutich. 2017a. The Phoenix Area Social Survey IV: Linking Social and Biophysical Dynamics in Urban Neighborhoods. A report from the Central Arizona–Phoenix Long-Term Ecological Research (CAP LTER) project. Tempe, Arizona.

Larson, K. L., and J. Brumand. 2014. Paradoxes in landscape management and water conservation: examining neighborhood norms and institutional forces. Cities and the Environment 7:6.

Larson, K. L., D. Casagrande, S. L. Harlan, and S. T. Yabiku. 2009. Residents' yard choices and rationales in a desert city: social priorities, ecological impacts, and decision tradeoffs. Environmental Management 44:921–937.

Larson, K. L., E. Cook, C. Strawhacker, and S. J. Hall. 2010. The influence of diverse values, ecological structure, and geographic context on residents' multifaceted landscaping decisions. Human Ecology 38:747–761.

Larson, K. L., J. Hoffman, and J. Ripplinger. 2017b. Legacy effects and landscape choices in a desert city. Landscape and Urban Planning 165:22–29.

Leong, M., R. R. Dunn, and M. D. Trautwein. 2018. Biodiversity and socioeconomics in the city: a review of the luxury effect. Biology Letters 14:20180082.

Locke, D. H., M. Avolio, T. Trammel, R. Roy Chowdhury, J. M. Grove, J. Rogan, D. G. Martin, N. Bettez, J. Cavender-Bares, P. M. Groffman, S. J. Hall, J. B. Heffernan, S. E. Hobbie, K. L. Larson, J. L. Morse, C. Neill, L. A. Ogden, J. P. M. O'Neil-Dunne, D. Pataki, W. D. Pearse, C. Polsky, and M. M. Wheeler. 2018. A multi-city comparison of front and backyard differences in plant species diversity and nitrogen cycling in residential landscapes. Landscape and Urban Planning 178:102–111.

Loram, A., J. Tratalos, P. H. Warren, and K. J. Gaston. 2007. Urban domestic gardens (X): the extent & structure of the resource in five major cities. Landscape Ecology 22:601–615.

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

Lu, J. W. T., E. S. Svendsen, L. K. Campbell, J. Greenfeld, J. Braden, K. L. King, and N. Falxa-Raymond. 2010. Biological, social, and urban design factors affecting young street tree mortality in New York City. Cities and the Environment 3:5.

Martin, C. A. 2008. Landscape sustainability in a Sonoran Desert city. Cities and the Environment 1:5.

Martin, C. A., K. A. Peterson, and L. B. Stabler. 2003. Residential landscaping in Phoenix, Arizona, U.S.: practices and preferences relative to covenants, codes, and restrictions. Journal of Arboriculture 29:9–17.

Martini, N. F., and K. C. Nelson. 2015. The role of knowledge in residential lawn management. Urban Ecosystems 18:1031–1047.

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

Namin, S., W. Xu, Y. Zhou, and K. Beyer. 2020. The legacy of the Home Owners' Loan Corporation and the political ecology of urban trees and air pollution in the United States. Social Science & Medicine 246:1112758.

Nassauer, J. I., Z. Wang, and E. Dayrell. 2009. What will the neighbors think? Cultural norms and ecological design. Landscape and Urban Planning 92:282–292.

Nelson, R. K., L. Winling, R. Marciano, and N. Connolly. 2020. Mapping inequality. American Panorama. https://dsl.richmond.edu/panorama/redlining/#loc=12/33.463/-112.152&city=phoenix-az.

Nesbitt, L., N. Hotte, S. Barron, J. Cowan, and S. R. J. Sheppard. 2017. The social and economic value of cultural ecosystem services provided by urban forests in North America: a review and suggestions for future research. Urban Forestry & Urban Greening 25:103–111.

NOAA. 2019a. Extreme temperature facts for Phoenix and Yuma. National Weather Service. https://www.wrh.noaa.gov/psr/climate/extremeTemps.php.

NOAA. 2019b. Data tools: 1981-2010 normals. National Climate Data Center. https://www.ncdc.noaa.gov/cdo-web/datatools/normals.

Nowak, D. J. 2012. Contrasting natural regeneration and tree planting in fourteen North American cities. Urban Forestry & Urban Greening 11:374–382.

Nowak, D. J., and E. J. Greenfield. 2012. Tree and impervious cover change in U.S. cities. Urban Forestry & Urban Greening 11:21–30.

Nowak, D. J., and E. J. Greenfield. 2018. Declining urban and community tree cover in the United States. Urban Forestry & Urban Greening 32:32–55.

Nowak, D. J., and J. T. Walton. 2005. Projected urban growth (2000-2050) and its estimated impact on the US forest resource. Journal of Forestry 103:383–389.

Ossola, A., D. Locke, B. Lin, and E. Minor. 2019. Greening in style: urban form, architecture and the structure of front and backyard vegetation. Landscape and Urban Planning 185:141–157.

Pearsall, H., and Z. Christman. 2012. Tree-lined lanes or vacant lots? Evaluating non-stationarity between urban greenness and socio-economic conditions in Philadelphia, Pennsylvania, USA at multiple scales. Applied Geography 35:257–264.

Pickett, S. T. A., M. L. Cadenasso, J. M. Grove, P. M. Groffman, L. E. Band, C. G. Boone, W. R. Burch, C. S. B. Grimmond, J. Hom, J. C. Jenkins, N. L. Law, C. H. Nilon, R. V. Pouyat, K. Szlavecz, P. S. Warren, and M. A. Wilson. 2008. Beyond urban legends: an emerging framework of urban ecology, as illustrated by the Baltimore Ecosystem Study. BioScience 58:139–150.

Pickett, S. T. A., M. L. Cadenasso, and S. J. Meiners. 2009. Ever since Clements: from succession to vegetation dynamics and understanding to intervention. Applied Vegetation Science 12:9–21.

Pincetl, S., T. W. Gillespie, D. E. Pataki, E. Porse, S. Jia, E. Kidera, N. Nobles, J. Rodriguez, and D. Choi. 2019. Evaluating the effects of turf-replacement programs in Los Angeles. Landscape and Urban Planning 185:210–221.

Polakowski, N. R., V. I. Lohr, and T. Cerny-Koenig. 2011. Survey of wholesale production nurseries indicates need for more education on the importance of plant species diversity. Arboriculture & Urban Forestry 37:259–264.

Pulsford, S. A., D. B. Lindenmayer, and D. A. Driscoll. 2016. A succession of theories: purging redundancy from disturbance theory. Biological Reviews 91:148–167.

R Core Team. 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ripplinger, J., S. L. Collins, A. M. York, and J. Franklin. 2017. Boom-bust economics and vegetation dynamics in a desert city: how strong is the link? Ecosphere 8: e01826.

Ripplinger, J., J. Franklin, and S. L. Collins. 2016. When the economic engine stalls - a multi-scale comparison of vegetation dynamics in pre- and post-recession Phoenix, Arizona, USA. Landscape and Urban Planning 153:140–148.

Robbins, P. 2007. Lawn people: how grasses, weeds, and chemicals make us who we are. Temple University Press, Philadelphia, Pennsylvania, USA.

Robinson, S. L., and J. T. Lundholm. 2012. Ecosystem services provided by urban spontaneous vegetation. Urban Ecosystems 15:545–557.

Rogers, D. A., T. P. Rooney, T. J. Hawbaker, V. C. Radeloff, and D. M. Waller. 2009. Paying the extinction debt in southern Wisconsin forest understories. Conservation Biology 23:1497–1506.

Roman, L. A., J. J. Battles, and J. R. McBride. 2014. Determinants of establishment survival for residential trees in Sacramento County, CA. Landscape and Urban Planning 129:22–31.

Roman, L. A., J. P. Fristensky, T. S. Eisenman, E. J. Greenfield, R. E. Lundgren, C. E. Cerwinka, D. A. Hewitt, and C. C. Welsh. 2017. Growing canopy on a college campus: understanding urban forest change through archival records and aerial photography. Environmental Management 60:1042–1061.

Roman, L. A., H. Pearsall, T. S. Eisenman, T. M. Conway, R. T. Fahey, S. Landry, J. Vogt, N. S. van Doorn, J. M. Grove, D. H. Locke, A. C. Bardekjian, J. J. Battles, M. L. Cadenasso, C. C. K. van den Bosch, M. Avolio, A. Berland, G. D. Jenerette, S. K. Mincey, D. E. Pataki, and C. Staudhammer. 2018. Human and biophysical legacies shape contemporary urban forests: a literature synthesis. Urban Forestry & Urban Greening 31:157–168.

Roman, L. A., and F. N. Scatena. 2011. Street tree survival rates: meta-analysis of previous studies and application to a field survey in Philadelphia, PA, USA. Urban Forestry & Urban Greening 10:269–274.

Roman, L. A., L. A. Walker, C. M. Martineau, D. J. Muffly, S. A. MacQueen, and W. Harris. 2015. Stewardship matters: case studies in establishment success of urban trees. Urban Forestry & Urban Greening 14:1174–1182.

Scharenbroch, B. C., J. E. Lloyd, and J. L. Johnson-Maynard. 2005. Distinguishing urban soils with physical, chemical, and biological properties. Pedobiologia 49:283–296.

Schroeder, P. J., and D. G. Jenkins. 2018. How robust are popular beta diversity indices to sampling error? Ecosphere 9: e02100.

Seto, K. C., M. Fragkias, B. Guneralp, M. K. Reilly, and B. Gu. 2011. A meta-analysis of global urban land expansion. PLoS ONE 6:1435–1439.

Smith, I. A., V. K. Dearborn, and L. R. Hutyra. 2019. Live fast, die young: accelerated growth, mortality, and turnover in urban street trees. PLoS ONE 14:e0215846.

The Plant List. 2013. Version 1.1. http://www.theplantlist.org/.

Tubby, K. V., and J. F. Webber. 2010. Pests and diseases threatening urban trees under a changing climate. Forestry 83:451–459.

Turner, V. K., and M. Stiller. 2020. How do homeowners associations regulate residential landscapes? An analysis of rule structure and content in Maricopa County (AZ). Journal of the American Planning Association 86:25–38.

U.S. Census Bureau. 2019a. New Census Bureau estimates show counties in South and West lead nation in population growth. https://www.census.gov/newsroom/press-releases/2019/estimates-county-metro.html

van Heezik, Y., C. Freeman, K. Davidson, and B. Lewis. 2020. Uptake and engagement of activities to promote native species in private gardens. Environmental Management 66:42–55.

Vogt, J. M., S. L. Watkins, S. K. Mincey, M. S. Patterson, and B. C. Fischer. 2015. Explaining planted-tree survival and growth in urban neighborhoods: a social-ecological approach to studying recently-planted trees in Indianapolis. Landscape and Urban Planning 136:130–143.

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

Wheeler, M. M., K. L. Larson, and R. Andrade. 2020. Attitudinal and structural drivers of preferred versus actual residential landscapes in a desert city. Urban Ecosystems 23:659–673.

Widney, S., B. C. Fischer, and J. Vogt. 2016. Tree mortality undercuts ability of tree-planting programs to provide benefits: results of a three-city study. Forests 7:65.

Yue, C., T. Hurley, and N. O. Anderson. 2012. Heterogeneous consumer preferences for native and invasive plants: evidence from experimental auctions. HortScience 47:1091–1095.

Ziter, C., R. A. Graves, and M. G. Turner. 2017. How do land-use legacies affect ecosystem services in United States cultural landscapes? Landscape Ecology 32:2205–2218.

Zmyslony, J., and D. Gagnon. 1998. Residential management of urban front-yard landscape: a random process? Landscape and Urban Planning 40:295–307.

TABLE 1. Neighborhood demographics from the 2010 U.S. Census and the 2017 American Community Survey for the most closely overlapping census tract for each sample neighborhood.

Year	Neighborhood	Dominant landscape type	Median household income ($)	Home age[1] (mean years ± s.d.)	% Owner occupied	% With children	% Residents with Bachelor's degree or higher	% Hispanic/ Latinx residents
2010	Old Hispanic Core	Mesic	38,000	58 ± 7	62	46	8	79
	Historic Palms District	Mesic	71,000	73 ± 7	74	13	54	16
	New Xeric Tracts	Xeric	56,000	17 ± 5	61	20	37	6
	Wealthy Mountain Oasis	Xeric	152,000	17 ± 2	95	55	65	3
2017	Old Hispanic Core	Mesic	35,000	64 ± 7	52	47	6	84
	Historic Palms District	Mesic	89,000	79 ± 7	74	15	56	20
	New Xeric Tracts	Xeric	63,000	23 ± 5	52	24	30	15
	Wealthy Mountain Oasis	Xeric	150,000	23 ± 2	94	35	61	8

[1] Calculated from 2018 tax assessor data on year built, for parcels in the 2018 vegetation sample

TABLE 2. Social and vegetation survey sample sizes by neighborhood.

Neighborhood	Social survey responses	Response rate (%)	Vegetation surveys 2008	2018	2019
Old Hispanic Core	6	5.9	106	105	6
Historic Palms District	39	40.6	101	95	39
New Xeric Tracts	29[1]	27.4	109	107	26
Wealthy Mountain Oasis	30	28.6	109	109	29
Unknown[2]	1				
Total	105	25.7	425	416	100

[1] Includes one respondent for whom vegetation data were excluded

[2] Unique identifier removed from completed survey

TABLE 3. Predictors of Bray-Curtis dissimilarity for yard vegetation communities from 2008 to 2018, including interactions between neighborhood and typology change.

Predictor	Estimate ± std. error	p-value
Intercept	0.40 ± 0.02	< 0.0001
Typology change occurred	0.19 ± 0.06	0.002
Home sold 2008-2018	0.12 ± 0.02	< 0.0001
Old Hispanic Core	0.29 ± 0.04	< 0.0001
Historic Palms District	0.16 ± 0.04	< 0.0001
New Xeric Tracts	0.07 ± 0.03	0.03
Old Hispanic Core × Typology change	-0.20 ± 0.07	0.008
Historic Palms District × Typology change	-0.10 ± 0.08	0.2
New Xeric Tracts × Typology change	-0.1 ± 0.1	0.3

Notes: Neighborhoods were compared against the Wealthy Mountain Oasis neighborhood. Model $R^2 = 0.20$, N = 415.

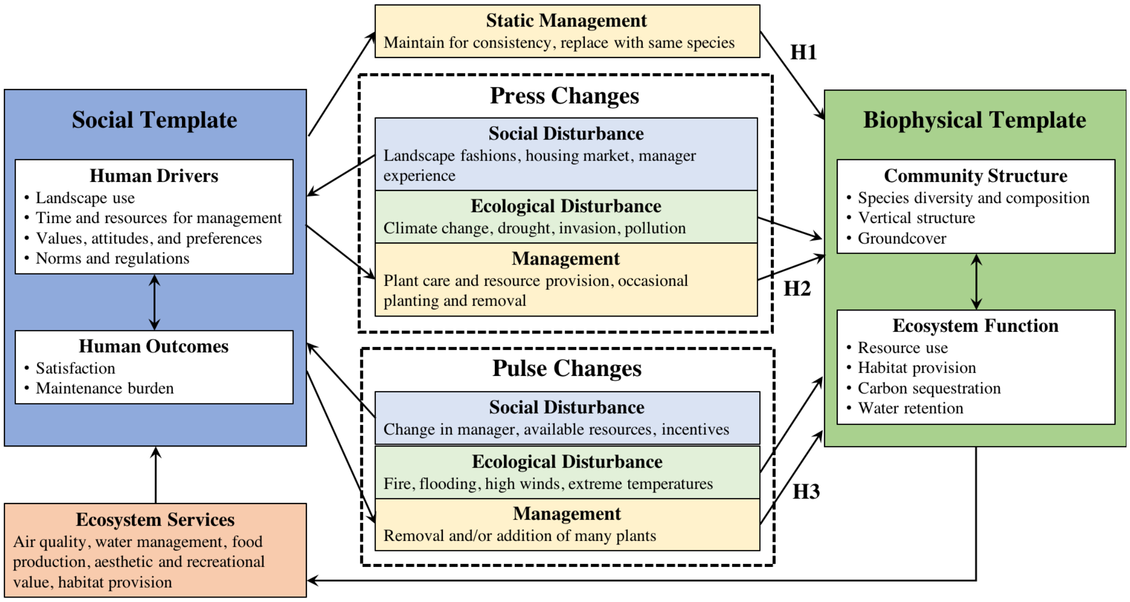

FIG. 1. A framework of managed urban vegetation change, modified from Collins et al. (2011) and Cook et al. (Cook et al. 2012). Hypotheses 1-3 as outlined in the text are shown as H1-H3.

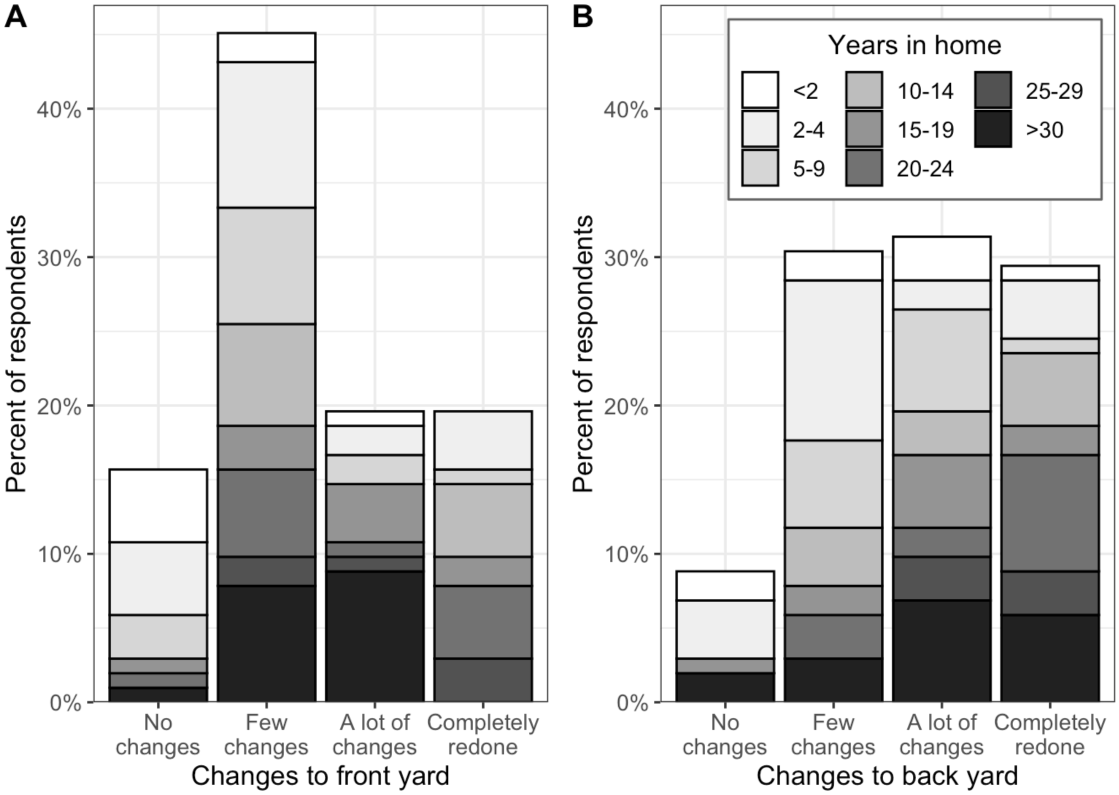

FIG. 2. Reported amount of change made to the (A) front yard and (B) back yard in the time the respondent had lived at their current home (shown by shades of gray). N = 102.

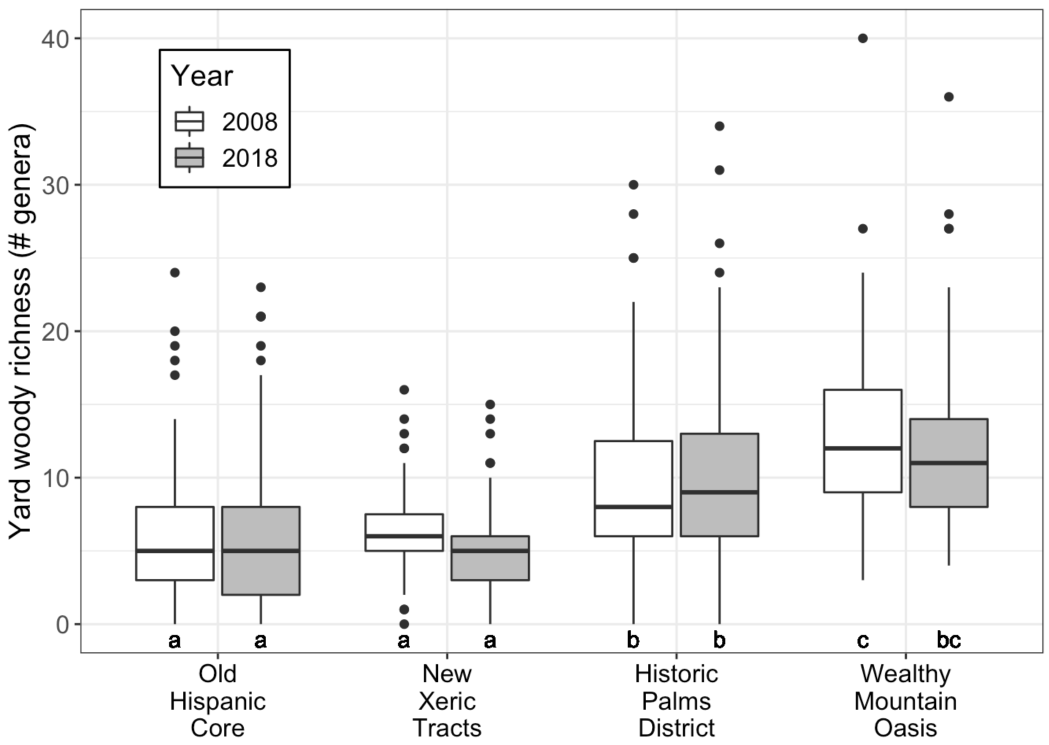

FIG. 3. Woody generic richness by neighborhood in 2008 and 2018. Annotations below boxes indicate significant differences in richness between neighborhoods and years, as determined by Tukey test following two-way ANOVA. Neighborhood median income increases along the x-axis (Table 1).

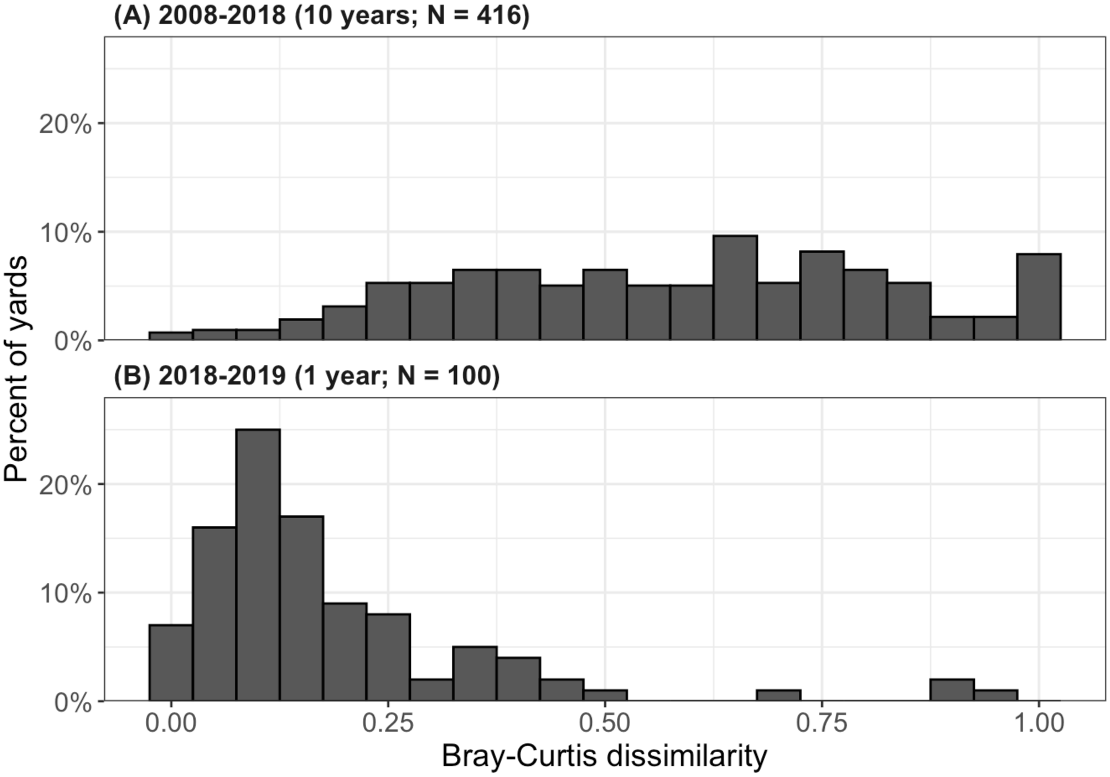

FIG. 4. Bray-Curtis dissimilarity of woody vegetation (trees, shrubs, vines, and succulents) at the genus level (A) in 416 yards from 2008 to 2018, and (B) in 100 yards from 2018 to 2019. Dissimilarity of 1 indicates complete turnover of the yard woody plant community. Jaccard dissimilarity (presence/absence only) shows similar patterns (Appendix D: Fig. S3).

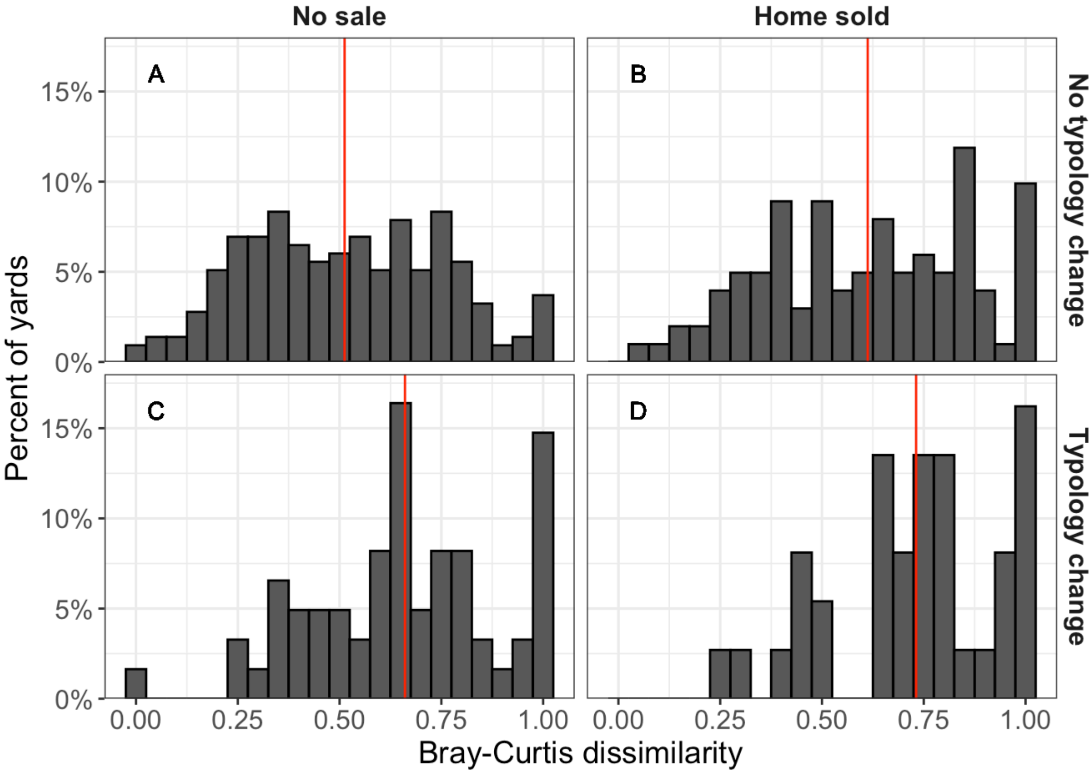

FIG. 5. Woody genus Bray-Curtis dissimilarity for front yards from 2008 to 2018 that have or have not changed ownership or typology from 2008 to 2018. Panels show yards that experienced (A) neither a change in ownership nor a typology change (N = 216); (B) change in ownership but not typology (N = 101); (C) change in typology but not ownership (N = 61); and (D) change in both ownership and typology (N = 37). Red lines mark mean dissimilarity values for each panel.

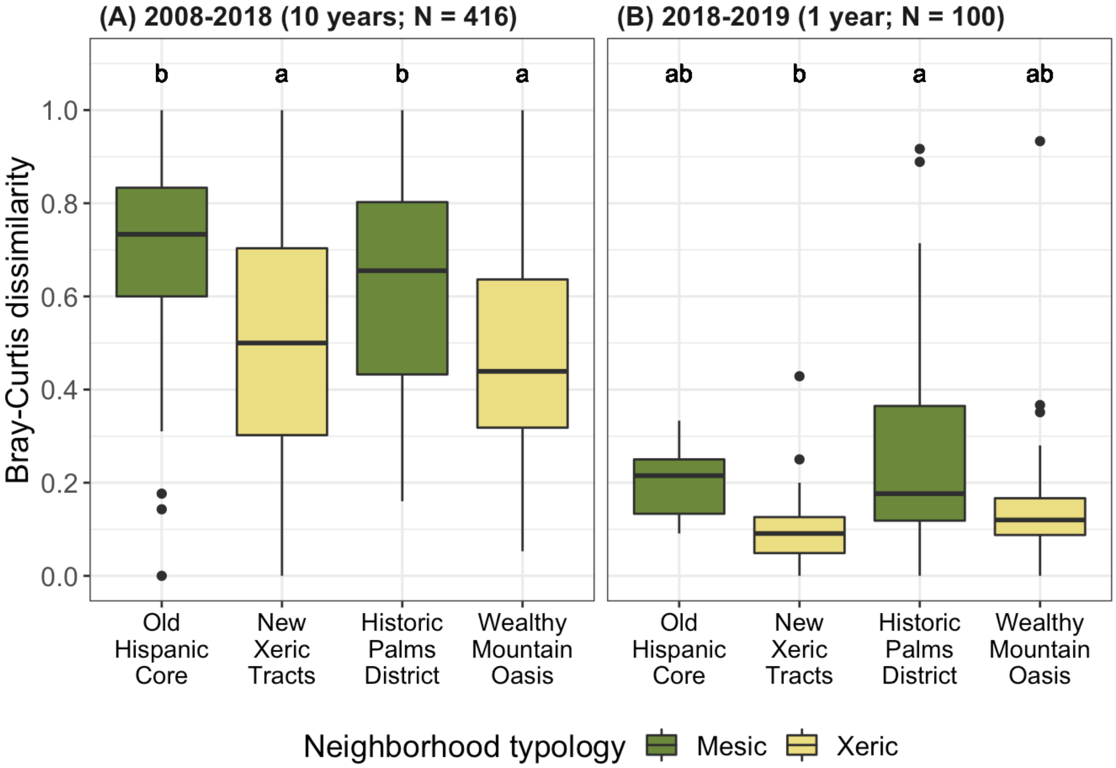

FIG. 6. Change in woody vegetation Bray-Curtis dissimilarity by neighborhood from (A) 2008 to 2018 and (B) 2018 to 2019. Annotations above boxes indicate significant differences in dissimilarity between neighborhoods, as determined by Tukey test following OLS modeling (Table 3). Box colors show the neighborhood dominant typology, either mesic (lawn-dominated) or xeric (desert-like). Neighborhood median income increases along the x axis (Table 1).

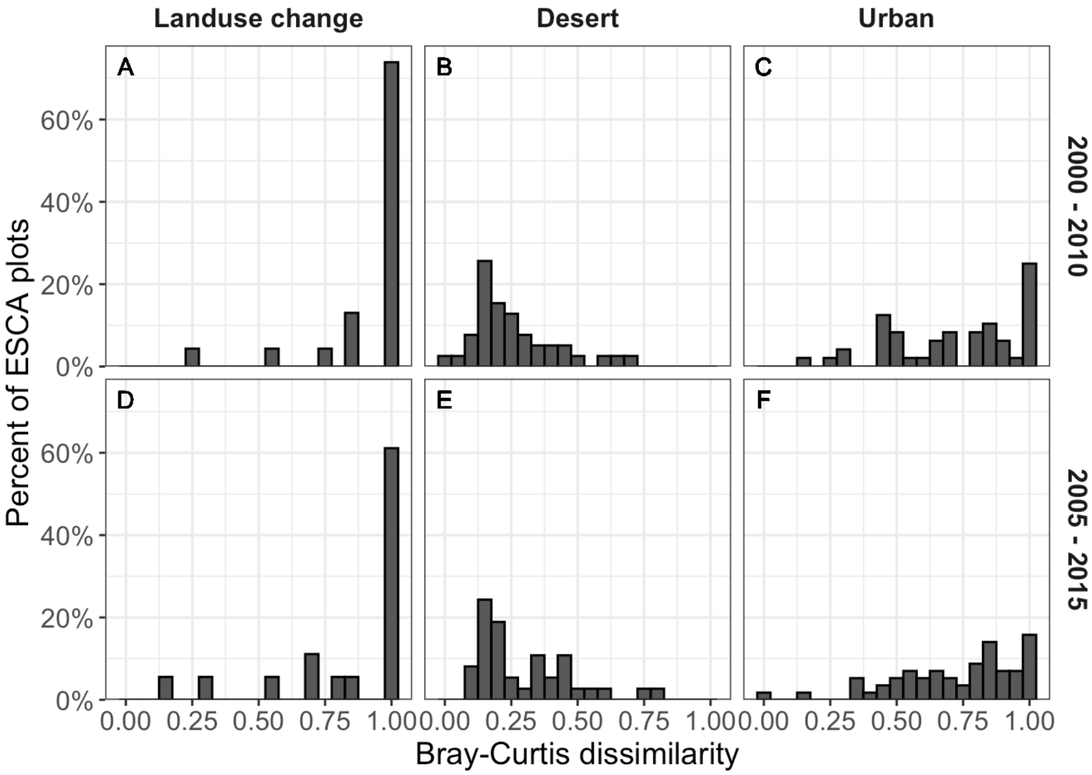

FIG. 7. Woody plant dissimilarity at the genus level between two sets of surveys done as part of the Ecological Survey of Central Arizona (ESCA). Panels show dissimilarity in (A) plots that changed dominant land use type from 2000 to 2010 (N = 23); (B) desert plots from 2000 to 2010 (N = 39); (C) urban plots from 2000 to 2010 (N = 48); (D) plots that changed dominant land use type from 2005 to 2015 (N = 18); (E) desert plots from 2005 to 2015 (N = 37); and (F) urban plots from 2005 to 2015 (N = 57). Dissimilarity values reflect changes at the genus level between the same plots over time. Distributions in (B) and (C) differ significantly, and distributions in (E) and (F) differ significantly (Two-sample Kolmogorov-Smirnov test, $p < 0.0001$).

CHAPTER 5

ENVIRONMENTAL ATTITUDES PREDICT NATIVE PLANT ABUNDANCE IN RESIDENTIAL YARDS

Abstract. Native plant landscaping can provide unique support for native wildlife in urban settings, but the drivers of native plant inclusion in private residential yards are not well characterized. As with other pro-environmental behaviors, native plant landscaping is likely driven by a combination of resident and landscape attributes. We ask, how do resident attitudes, knowledge, plant choice priorities, demographics, and parcel structure predict existing native plant abundance? To address this question, we compared resident characteristics with front yard vegetation in 105 parcels in Phoenix, Arizona. Although many residents had positive attitudes toward native plants, less than a third of woody plants in most yards were native. Native plant abundance was higher in xeric rock-covered yards where residents believed native plants belonged in the city, prioritized choosing native plants, and had higher household income. Reported knowledge about native plants was low, but did not predict native plant abundance. Although native plants in the arid environment of Phoenix are adapted to low water conditions, residents who prioritized low water use plant selection had fewer native plants, highlighting an opportunity for native plant marketing. These results suggest that educational campaigns to increase resident knowledge of native plant identification and care are unlikely to result in greater native plant abundance in the residential landscape. Marketing native plants to highlight qualities such as low water needs and addressing barriers such as horticultural availability and expense should be further investigated as potential methods of increasing native plant resources in urban environments.

INTRODUCTION

Residential yards are a dominant feature of cities, with 92% of new U.S. homes including some outdoor space (U.S. Census Bureau 2019b). These urban spaces can support diverse wildlife communities, depending on land management practices (Goddard et al. 2009, Aronson et al. 2017, Derby Lewis et al. 2019). Landscaping with native species can support biodiversity by providing resources for wildlife that are unmatched by horticultural nonnatives (Burghardt et al. 2009, Pardee and Philpott 2014, Threlfall et al. 2017, Narango et al. 2018). As more people move to cities, using native vegetation to create locally distinct urban landscapes can also support connection to the environment and sense of place (Hooper et al. 2008). Emphasis on native plants in landscaping can help prevent urban homogenization, in which cities host the same sets of species across broad geographic regions, thus reducing the ability of cities to support local native wildlife and flattening local environmental distinctiveness (McKinney 2006, Groffman et al. 2014). Landscaping with native plants is one way for urban residents to improve the conservation value and unique character of cities.

Conservation action, such as landscaping with native plants, is determined by human behavior and decision-making (Schultz 2011). In yards, individual residents make choices about management actions to take, which are guided by personal motivations as well as the social and physical environment (Cook et al. 2012). These choices can have important outcomes for urban biodiversity (Goddard et al. 2013, Belaire et al. 2016). Environmental attitudes sometimes predict behavior and thus environmental outcomes, but social, financial, and other constraints often prevent direct correspondence between attitudes and behavior (Heberlein 2012). Few studies have investigated how resident attitudes relate to native plant outcomes, despite the ecological benefits of native plants. Here, we use a case study in the arid city of Phoenix, Arizona, to ask, how do attitudinal drivers, priorities, and physical yard characteristics predict native plant abundance in residential yards?

Native plants in urban landscapes

Native plants have long been a subject of interest in horticulture and urban landscape design, though they are often perceived as a relatively small niche market (Potts et al. 2002, Hooper et al. 2008, Kauth and Pérez 2011). Horticultural industry professionals have identified several motivations driving native plant sales in urban landscapes, including reduced yard maintenance requirements, limited water use, and habitat provision for wildlife (Potts et al. 2002, Hooper et al. 2008, Brzuszek and Harkess 2009). Similarly, participants in the U.S. Master Gardener program reportedly selected native plants due to their adaptations to the local environment (Brzuszek et al. 2010). Landscape designers in the southwestern U.S. reported increasing the use of native plants in their work over the previous five years, though very few considered themselves to be expert native plant users (Hooper et al. 2008). Moreover, designers perceive a lack of public support and enthusiasm for native landscaping (Crewe 2013). Despite the benefits of native plant cultivation, public attitudes toward native plants are unclear, as is the extent to which attitudes and other factors influence the abundance of native plants in residential landscapes.

The term "native plants" is often used in both gardening and ecology, but its definition can vary when put into practice. Ecologically, native species are defined as organisms that have a long evolutionary history in a particular location, likely with coevolved species and adaptations suitable to local environmental conditions. However, nativity is not a trait that can be measured, so classifications of individual species are typically based on judgements of how long a species has been in a location and how it arrived (Kendle and Rose 2000). Further, nativity classifications are regularly made following political rather than geographic boundaries, resulting in different statuses across ecologically meaningless divisions.

While ecologists use geopolitical and historical nativity classifications to describe whether a species is appropriate in a given location, non-ecologists may think of species with cultural connections to a city or large urban populations as belonging there, regardless of ecological nativity (Head and Muir 2004, 2006). Thus, plants seen as belonging, and sometimes colloquially considered native, could include those with a long history in the local urban environment, adaptations that make it well suited to the local environment, or historic representations in the local culture. For example, in arid environments in Australia, some residents conflate the ideas of nativity and drought-tolerance (Head and Muir 2004), although many species from other regions are also adapted to limited water availability. While ecologists and horticultural professionals may attach importance to nativity due to the roles native species play in healthy ecosystems, members of the general public may take a more pragmatic, functional, or culturally based approach to landscaping without regard to a plant's status as native or not.

Although native plants are adapted to the local environment, they may be poorly suited to urban heat, pollution, and disturbance, and so may not always be an appropriate choice for landscaping (Kendle and Rose 2000). Native plants can also be difficult for nurseries and landscapers to source and propagate, which partially explains why only about a quarter of native vascular plant species in the U.S. are available commercially (Potts et al. 2002, White et al. 2018). Thus, finding appropriate native species at local nurseries and big box stores can be a challenge for consumers. Further, horticultural professionals suggest that consumer education is necessary for proper maintenance and to manage expectations for native plants (Potts et al. 2002, Brzuszek et al. 2007, Hooper et al. 2008, Brzuszek and Harkess 2009, Kauth and Pérez 2011, Crewe 2013). While native plants can provide benefits for biodiversity and place attachment, they may present important functional challenges for residents and landscapers.

Environmental attitudes, knowledge, and native plants

Environmental attitudes can influence pro-environmental behaviors, such as the decision to landscape with native plants (Kollmuss and Agyeman 2002, Schultz 2011, Heberlein 2012). Simply defined, attitudes are positive and negative judgements about some object or phenomenon such as native plants (Larson 2010, Heberlein 2012). While attitudes do not always predict behavior, they can reveal motivations and constraints around actions and thus are important to promote public support and desirable behaviors (e.g., planting natives). Changing people's attitudes is rarely an effective way to increase pro-environmental behaviors, but understanding attitudes and working with existing motivations is an effective strategy to promote conservation goals (Schultz 2011, Heberlein 2012).

Knowledge has sometimes been linked with environmental behavior, including yard management (Frick et al. 2004, Levine and Strube 2012, van Heezik et al. 2012, Martini and Nelson 2015). Lack of knowledge about native plants has been specifically implicated as a major barrier to their use in residential landscaping (Potts et al. 2002, Hooper et al. 2008, Brzuszek and Harkess 2009, Kauth and Pérez 2011). For example, customer unfamiliarity and confusion over what native wildflowers are have been identified as two major limitations to their adoption (Kauth and Pérez 2011). A study of Australian residents who converted their yards from typical English-style gardens to native themes identified knowledge of environmental issues as a key driver of the choice to convert yards (Uren et al. 2015). In the UK, people were able to identify when landscaping was primarily native but did not prefer natives (Hoyle et al. 2017), suggesting that knowledge is not the primary barrier to inclusion everywhere. Further, a survey in Europe suggested that nativity was not an important concept to the public when evaluating the desirability of a species (Fischer et al. 2011), and therefore was unlikely to affect landscaping choices. These results suggest that both knowledge and attitudes toward native plants play a part in choices to include native vegetation in yard landscaping.

Typically, more specific attitudes and those with greater relevance to a particular attitude object, such as native plants, are more predictive of behavior than are more general attitudes (Kim and Hunter 1993). In residential yards, specific attitudes toward particular plant features can drive management decisions (Kendal et al. 2012). Similarly, specific attitudes toward low water use plants or plants with thorns may be more predictive of succulent abundance than general environmental attitudes. However, more general attitudes may also be relevant, as in Australia where attitudes toward native bush habitat overall are related to ideas of naturalness, cultural connection, and beliefs about whether different native and non-native plants belong in the urban landscape (Head and Muir 2004, 2006). Therefore, we consider both more specific attitudes toward native plants and more general attitudes toward the regional desert environment as potential predictors of native plant landscaping.

Previous research in the U.S. has identified consistent key priorities (i.e., important considerations) for residential yard management, including low maintenance requirements, neat and orderly appearances, and aesthetic beauty (Larson et al. 2009, Nassauer et al. 2009, Cook et al. 2012). In accordance with these priorities, choices to purchase and install native plants in landscaping may be linked to plant traits (Kendal et al. 2012). In Minnesota, residents who prioritized supporting wildlife and creating a beautiful yard cultivated a greater diversity of native plants, showing connections between native planting and yard management priorities in a humid continental climate (Cavender-Bares et al. 2020). In the arid southwestern U.S., residents reported positive attitudes toward desert plants, including natives, mostly agreeing that they look attractive and provide sufficient variety (Spinti et al. 2004, St. Hilaire et al. 2010). Additionally, residents reported that they would use native plants if they conserved water and if they were attractive (Lockett et al. 2002). However, residents have also expressed concerns about the presence of thorns on plants in yards where pets and children may play (Larson et al. 2009). Although informative, these studies have not tested whether prioritizing traits such as beauty, maintenance requirements, thorns, or nativity align with actual native plant cultivation in residential yards.

Beyond attitudes: resident characteristics and urban structure

Attitudes alone cannot predict behavior, and the gap between attitudes and behavior can sometimes be explained by social characteristics and urban structure (Kollmuss and Agyeman 2002). For residential yards, studies have shown more abundant and diverse vegetation in higher income neighborhoods (Hope et al. 2003, Cook et al. 2012, Avolio et al. 2018, Gerrish and Watkins 2018). One proposed mechanism for this so-called "luxury effect" is financial resources (Hope et al. 2003), which might be particularly relevant for native plants because they are often more expensive than non-natives (Brzuszek et al. 2007, Avolio et al. 2018). Education level can also predict yard vegetation outcomes, such as tree planting rates and cultivated plant composition (Roman et al. 2014, Padullés Cubino et al. 2018). The local specificity of native plants suggests that acculturation to a particular region might affect attitudes toward native plants, and thus related behaviors. For example, in Phoenix, Arizona, newer residents tend to prefer naturalistic xeric landscaping while long-term residents prefer grass (Martin et al. 2003, Wheeler et al. 2020). By extension, newcomers may also embrace native plants relative to long-term residents.

Urban structure, or parcel and neighborhood characteristics, also affect yard outcomes. Lot and garden size can constrain the vegetation possibilities by affecting the pervious area, managed area, and amount of vegetation present (Bigsby et al. 2014, Ossola et al. 2019). Vegetation management may also be driven by a desire to match home and yard aesthetics, such as an adobe-style house with a desert-like yard and desert species, or a brick colonial house with a manicured English-style garden (Peterson et al. 2012, Uren et al. 2015, Ossola et al. 2019). Native species may be perceived as not fitting in with a manicured yard aesthetic (e.g., turfgrass lawns), or may be seen as the most appropriate choice for a yard with a naturalistic design (e.g., gravel groundcover in desert regions).

Research aims and hypotheses

In this research, we evaluate how resident attitudes and priorities, demographic characteristics, and parcel structure are associated with the abundance of native plants in residential yards. While related research has been conducted on the ecological benefits of native plants (e.g., Narango et al. 2018), industry professional perceptions of native plants (e.g, Potts et al. 2002), and the adoption of water-conserving landscapes (e.g., Spinti et al. 2004), little work has yet evaluated how diverse drivers affect native plant abundance in cities. Using a paired social and vegetation survey of residential yards in Phoenix, Arizona, we tested four hypotheses:

H1) Plant nativity is a recognizable and important trait for residents.

H2) Plant selection priorities will best predict native plant abundance, followed by attitudes toward native plants, more general attitudes toward the desert, and resident knowledge of native plants.

H3) Native plants are selected for landscaping when residents prioritize low water use, low maintenance needs, and providing habitat for wildlife, but are avoided due to negative aesthetic perceptions, potential hazards (e.g., cactus spines), and lack of availability or expense of purchasing.

H4) Resident characteristics and parcel structure have more influence on native plant abundance than do resident attitudes and priorities.

To address these hypotheses, we explored variation in front yard native plant abundance using resident knowledge of native plants, attitudes toward native plants, attitudes toward the desert, plant selection priorities, resident characteristics, and parcel structure as predictors to determine their relative importance.

METHODS

Study location

We conducted our study in the city of Phoenix, Arizona, which is located in the Sonoran Desert of the southwestern United States. Historically, Phoenix has been viewed and advertised as a desert oasis, in which the warm climate is celebrated but the desert is seen as separate from the city and as a challenge to be conquered by urban planning and design (Zube et al. 1986). To fit this vision, residential landscaping has traditionally been lush and grassy (Zube et al. 1986). However, new developments are more often landscaped with desert-like xeric designs reminiscent of the local context, particularly as developers are required to plan for future water security (Heavenrich and Hall 2016, Frost 2016). Xeric yards are typically a mix of rock groundcover with some drought-adapted plants and trees, but no grass. Landscape architects in Phoenix have increasingly incorporated and highlighted native plants in their designs and have made advances in identifying suitable species, but they have faced a lack of public support (Crewe 2013).

Residential yard sampling

We conducted paired vegetation and social surveys to characterize the plant composition of residential yards and motivations of residents. Our study yards were chosen as part of a long-term sampling effort (Larson et al. 2010, Larson and Brumand 2014). Four focal neighborhoods were defined by 2000 U.S. Census block groups and are arranged roughly along a north-to-south transect in the city of Phoenix. The neighborhoods represent a range of socioeconomic characteristics and include two older, primarily grassy (mesic) neighborhoods and two newer, primarily xeric neighborhoods (Table 1; see Chapter 4 for additional neighborhood description). Approximately 100 parcels were randomly selected in each neighborhood when the study originated in 2008. We carried out paired vegetation and social surveys at 416 yards from these neighborhoods in the summer of 2018 (Table 1).

Front yard vegetation

To quantify yard vegetation, we conducted visual surveys from the front sidewalk and identified all woody vegetation in the front yards of 416 focal parcels. We did not enter the yards unless invited by the resident. We identified plants to the lowest possible taxa using visible morphological traits and knowledge of the local horticulturally available species. We included only woody plants (trees, shrubs, succulents, and woody vines) in this analysis, as herbaceous and grass species could not be reliably identified from sidewalk surveys. In each yard, we recorded the number of individuals and growth form (tree, shrubs, succulent, or vine) for each taxon. We then calculated the number of native woody individuals and native woody species richness in each front yard.

We classified all taxa by nativity in two ways: first, nativity to the Sonoran Desert, which included southern Arizona and parts of northern Mexico; and second, nativity to only the Arizona Sonoran Desert. We expected those species native to the Arizona Sonoran Desert would more closely match what Phoenix residents perceive as native to the local region, while the Sonoran Desert in its entirety more closely matches habitat suitability-based descriptions of nativity that would be more often used in ecological applications.

Species were classified as native or non-native to the Arizona Sonoran Desert using the USDA Plants database, Biota of North America Program database (BONAP) and a cultivated plant encyclopedia (Bailey and Bailey 1976, Kartesz 2015, USDA NRCS 2020). Species with county-level native distributions in the Arizona Sonoran Desert region were classified as native to both the Arizona Sonoran Desert and Sonoran Desert. Species that were not native to the Arizona Sonoran Desert were compared against a checklist of native Mexican flora (Villaseñor 2016) to assess nativity in Sonoran regions of Mexico (see Appendix E for complete classification methods). Not all individuals could be identified to species using our sidewalk survey method (33% of individuals identified to genus only, 3% not identifiable to genus). These individuals were classified as native if all species recorded in the region were native, or if the most common species of the genus recorded in other Phoenix urban flora surveys was native (Appendix F: Table S1). Five common genera in the sample were classified as native to the Arizona Sonoran Desert (*Agave* sp., *Ferocactus* sp., *Leucophyllum* sp., *Optunia* sp., and *Yucca* sp.), although many individuals were horticultural hybrids of unclear origin. These genera are all desert-adapted and contain common species native to the Sonoran Desert. Of the 3% of species not identifiable to genus, we classified unidentified barrel, hedgehog, and columnar cacti as native to the Arizona Sonoran Desert, and unidentifiable broadleaf trees, broadleaf shrubs, and hedge-type shrubs as non-native. With this approach to defining nativity, we likely included some non-native individuals as native, but are unlikely to have misclassified many native individuals due to our broad inclusion of unidentified species as native. Thus, our quantification of native plants is at the upper bound of existing conditions.

We focus our results primarily on native plant abundance, with the assumption that a greater abundance of native plants would provide greater associated services, such as resources for native wildlife. Previous studies in residential areas have supported the relationship between native plant abundance and native wildlife using metrics such as native plant biomass (Narango et al. 2018) and native plant cover (Pardee and Philpott 2014). Further, in our sample of 416 yards, native plant abundance was correlated with native species richness (Pearson correlation = 0.84).

Resident attitudes, knowledge, and characteristics

During the summer of 2018, we mailed surveys assessing resident attitudes and motivations to 425 Phoenix households, including the 416 for which we obtained vegetation data. Residents were initially contacted via postcard to alert them to the study, and then were mailed a printed survey with a stamped and addressed return envelope. A reminder postcard was sent following survey mailing, and 15 $25 Visa gift cards were raffled among respondents as an incentive for participation. All survey materials were available in Spanish by request. Survey materials were approved by the ASU Institutional Review Board (Appendix C).

A total of 105 surveys were returned, yielding a response rate of 25.7% (Table 1). However, responses included one for which no vegetation data were available and one for which the unique identifier was removed, prohibiting linkage with vegetation data. Responses were unevenly distributed by neighborhood, with few respondents in the Old Hispanic Core (5.9% response rate) resulting in bias toward higher income, more educated, white homeowners. Previous research in this neighborhood has yielded similarly low engagement (Larson et al. 2017a). The average respondent was 56 years of age (range 23-91) and had lived in their current home for 16 years (range 0 - 58 years). Nearly all respondents owned their home (94%) and self-identified as white (94%). Most respondents were female (67%) and reported higher education (77% with bachelor's degree or higher) and income (median $120,000-$160,000, 38% over $200,000) than the neighborhood average based on U.S. Census data. Thus, our findings should be generalized with caution, and future research should endeavor to more fully explore the perspectives and landscapes of diverse residents.

To assess resident knowledge of and attitudes toward native plants, we first prompted survey respondents to think of native plants as "trees or other plants that come from or grow naturally in the desert around Phoenix. Do not include plants that come from other regions or parts of the world." Self-reported knowledge of native plants was evaluated with two statements: "I know a lot about gardening with native plants," and "I know how to determine whether a plant is native." Participants responded on a five-point scale from strongly disagree (1) through neither agree nor disagree (3) to strongly agree (5). We averaged these two responses addressing different facets of knowledge about native plants into a single native plant knowledge scale (Pearson correlation = 0.62; Table 2). We also asked residents approximately how many of the plants in their front yard were native to the Phoenix region, with the options none, a few, most, or all of my front yard plants are native, and an additional "not sure" choice.

We considered resident attitudes toward a more general attitude object (the desert) and a more specific one (native plants). Attitudes toward the desert were measured with a previously-used scale for this region (Andrade et al. 2019, Wheeler et al. 2020). Respondents were asked how strongly they agreed or disagreed with four statements: "the desert is an empty wasteland," "the desert is a very special place to me," "the desert is beautiful," and "the desert is a nice place to spend time." Responses were averaged with the first (negatively worded) statement reversed to give a unidirectional scale of attitudes toward the desert (Cronbach's alpha = 0.89; Table 2). For specific attitudes toward native plants, we asked how strongly respondents agreed or disagreed with two statements: "native plants do not belong in the city" and "native plants are beautiful." These items were not correlated (Pearson correlation = -0.07) and were both retained as independent attitudes toward native plants. Perception of native plants as beautiful was correlated with attitudes toward the desert (Pearson correlation = 0.56), but was unique enough that we included both in analyses.

Similar to previous work addressing value-based motivations for yard vegetation management (Kendal et al. 2012, Larson et al. 2016, Padullés Cubino et al. 2020), we asked residents to rank the importance of several priorities in their choice of new trees or other plants for their yard. If a respondent had not selected trees or other plants for their yard in the past, they were instructed to answer as if they were going to plant something now. Response options were on a four-point scale: (1) not at all important, (2) slightly important, (3) important, and (4) very important. We investigated seven different priorities: whether the tree or other plant is *easy to get*, whether the tree or other plant is *native* (grows naturally in the desert around Phoenix), whether the tree or other plant needs a lot of *water*, whether the tree or other plant will *attract birds or other wildlife*, whether the tree or other plant is *beautiful*, whether the tree or other plant has spines or *thorns*, and whether the tree or other plant is *low maintenance* (does not require much trimming; Table 2).

To address personal characteristics that may affect native plant adoption, we asked respondents about income, education, and how long they had lived in Phoenix. For income, we asked about the 2017 total combined income before taxes for all people in the household using response options in $20,000 increments up to $200,000. We also included an "over $200,000" response and a "prefer not to say" response. Income was treated as an ordinal variable from 1 to 11, with prefer not to say responses omitted (n = 13). For education, we asked, what is the highest level of school you have had a chance to complete? Responses were (1) less than high school, (2) high school, (3) community college, vocational school, or trade school, (4) bachelor's degree, and (5) graduate or professional school. Finally, we asked in what year the respondent was born and for how many years they had lived in the Phoenix Valley. Following Larson et al. (2016), we divided the number of years lived in Phoenix by the respondent's age in 2018 to get the percentage of life lived in Phoenix as a measure of local acculturation.

Parcel structure

In addition to resident characteristics, we considered two aspects of parcel physical structure: front yard area and rock cover. Front yard area was calculated in ArcGIS in 2008 by matching parcel boundary shapefiles from the Maricopa County tax assessor records with 2005 aerial photos (0.3m resolution) and 2009 satellite images from Google Maps. Front yards were manually outlined to calculate yard area. Approximate percent rock cover of the yard was used as a metric for the "xeric-ness" of the landscape aesthetic, with the idea that more xeric landscaping may have more native plants. During vegetation surveys, we divided each front yard into four quadrants and visually estimated the percent cover of rock in each quadrant. The estimates were then averaged to get overall yard percent rock cover.

Models of native plant abundance

We ran generalized linear models to test the effects of resident knowledge, attitudes, characteristics, and parcel structure on front yard native plant abundance and species richness. A total of 80 parcels had data for vegetation and all drivers included in the models, out of 103 parcels with returned social surveys and completed vegetation surveys. All analyses were run in R version 3.6.1 (R Core Team 2019).

We built four abundance models with the same set of predictors and different response variables to test for differences in the drivers of: (1) Sonoran Desert native plants, (2) Arizona Sonoran native plants, (3) all succulents, and (4) all woody plants. We also modeled Arizona Sonoran Desert native species richness to compare the drivers of abundance with drivers of diversity. For abundance, we considered both definitions of nativity to see if locally abundant species (i.e., Arizona Sonoran natives) more closely matched with resident priorities than did regionally native species (i.e., Sonoran natives). We also considered abundance of all succulents to represent a group of species with similar adaptations to low-water environments but of varying nativities. The comparison with total woody plant abundance tested whether the drivers of native plant abundance could be explained more simply as drivers of overall abundance, where native plants increase as a constant proportion of total plants.

For each response variable, we built a generalized linear model with a Poisson distribution, which was run using R function glm. Model predictors were the native plant knowledge index, general attitudes toward the desert, specific attitudes toward native plants (beautiful and belonging in the city), priorities for plant selection (native, low water use, easy to get, beautiful, attract wildlife, based on spines or thorns, and low maintenance), resident characteristics (income, education, and percent life in Phoenix), and parcel structure (front yard area and front yard rock cover; Table 2). Pairwise correlations for all predictors were less than 0.6 and variance inflation factors were less than 4. To select the best models for each response variable, we conducted stepwise selection using AIC, starting with each global model and using the stepAIC function from package MASS version 7.3-51.4 (Ripley et al. 2019). Finally, we calculated standardized beta values to compare the relative importance of each predictor using the lm.beta function from package lm.beta version 1.5-1 (Behrendt 2014).

RESULTS

Native plant abundance and diversity

We observed 8,219 individual woody plants in 416 yards, of which 19% were native to the Arizona Sonoran Desert, 11% were native to the Sonoran Desert but not Arizona, 70% were non-native, and <1% could not be classified. On average, yards contained two species and four individuals that were native to the Arizona Sonoran Desert, and three species and six individuals native to the Sonoran Desert more broadly (Fig. 1, 2). Yards had 20% native individuals and 22% native species on average (out of all woody individuals and species, respectively) for native defined as the Arizona Sonoran Desert (31% native individuals and species for Sonoran Desert).

Importance and relevance of plant nativity (H1)

Survey respondents rated nativity as an important consideration when selecting new plants, with 68% rating it as important or very important and only 5% rating nativity as not important (Fig. 3). Low water use was the top priority for respondents, with 86% rating it as important or very important. Choosing beautiful and low maintenance plants were also more highly ranked priorities than nativity. Attitudes toward native plants were positive overall, with most respondents agreeing that native plants were beautiful and disagreeing that they don't belong in the city (i.e., believe they do belong; Table 2).

While a majority of respondents agreed that nativity was an important trait, they were mostly neutral in their reported knowledge of native plants, neither agreeing nor disagreeing that they knew how to determine nativity or how to garden with native plants (Table 2). In addition, 25% of respondents reported that they were not sure what proportion of plants in their front yard was native. Where respondents did report how many of their front yard plants were native, their classifications did not closely match our definitions of nativity (Fig. 4). To better understand this lack of connection, we looked at the community composition of yards for which the resident reported having all native plants. In 12 of these 13 yards, our surveys found that less than half of woody plants were native (Fig. 4). We saw a range of plants represented in these 12 yards, including drought-tolerant, non-native shrubs that are commonly cultivated in the region (*Lantana* sp. in 7 yards, *Bougainvillea* sp. in 4 yards, *Nerium oleander* in 3 yards), commonly cultivated non-native succulents (*Echinocactus grusonii* in 5 yards, *Aloe* sp. in 3 yards), and species native to the Sonoran Desert broadly but not the Arizona Sonoran Desert (*Hesperaloe parviflora*, 6 yards, *Leucophyllum* sp., 5 yards). Several of these yards also contained an iconic native species, with six containing either *Carnegiea gigantea* (saguaro) or *Fouquieria splendens* (ocotillo).

Predictors of native plant landscaping (H2-H4)

Plant choice priorities, resident attitudes and characteristics, and parcel structure significantly predicted front yard native plant abundance (Table 3). Comparing standardized beta values, the drivers with the greatest influence on native plant abundance were beliefs that native plants are beautiful and belong in the city, priorities for choosing native and low water use plants, household income, and yard rock cover. As expected, respondents who prioritized choosing native plants and believed that they belong in the city had more in their yards. However, prioritizing natives did not predict greater native plant species richness (Table 4). Contrary to expectations, residents for whom water use was an important consideration had significantly fewer native plants and lower native plant species richness. Additionally, residents who agreed that native plants were beautiful actually had fewer individuals and species in their yards. Prioritizing plants that attract wildlife and choosing plants based on spines and thorns both predicted greater native plant abundance, though with smaller effects than other predictors. Knowledge of native plants did not predict native plant abundance or species richness. Attitudes toward the desert were positively associated with native plant abundance depending on the definition of nativity used (Table 4).

There were few differences in predictors between the two definitions of nativity. Attitudes toward the desert were significantly positively related to Sonoran native abundance, but not Arizona Sonoran native abundance. Prioritizing easy to get plants and plants based on thorns or spines were negatively related to Arizona Sonoran native abundance and education was positively related to Arizona Sonoran native abundance, though none of these predictors were statistically significant. The model variables explained more variance in Arizona Sonoran native plants than in Sonoran native plants (Table 3), supporting the idea that Arizona Sonoran is the more appropriate definition of nativity.

While drivers were similar for both definitions of nativity, they differed for succulent abundance and total woody plant abundance. Importance of native and low water use plants did not predict abundance of succulents as it did native plants, despite succulents sharing adaptations for low water use. Respondents who prioritized low water requirements had slightly more woody plants overall. Residents who wanted low maintenance plants had fewer plants total, but also had fewer succulents, which typically have low maintenance requirements. Similarly, those who prioritized plants that are easy to get had slightly fewer native plants, succulents, and total woody plants, although this effect was small and not always statistically significant. Spines and thorns did not appear to be a detractor for either native plants or succulents. As expected, residents who wanted to attract wildlife had significantly more native plants, succulents, and woody plants.

Prioritizing beautiful plants predicted more succulents and woody plants and did not negatively impact native plants as we might expect if residents believed native plants were not beautiful. The importance of choosing beautiful plants did predict decreased native plant species richness (Table 4).

Households with higher income had significantly more native plants, succulents, and woody plants (Table 3). Income was the most important predictor of total woody plant abundance, but was equally weighted with other predictors for native plants and succulents. Duration of residence in the Phoenix Valley was positively associated with total woody plant abundance, but not significantly related to succulent or native plant abundance.

Front yard rock cover, a proxy for extent of xeric landscaping, was positively associated with native, succulent, and total woody plant abundance and with native plant species richness. The effect of rock cover on total woody plant abundance was small relative to other drivers, while its effect on native and succulent abundance was greater. Larger yards contained more woody plants but slightly fewer succulents.

DISCUSSION

Nativity is important, but knowledge is low (H1)

Phoenix residents recognized nativity as an important plant characteristic, but most reported a lack of knowledge about native plants and less than a third of plants in most front yards were native. Thus, we conclude that while nativity is a relevant concept for these residents, other factors are more important in structuring yard composition. As suspected by horticultural professionals, we found a knowledge gap surrounding the cultivation of native plants in yards (Potts et al. 2002, Brzuszek and Harkess 2009, Kauth and Pérez 2011). Many residents did not know whether their front yard plants were native, and of those who did, most assessments did not match our classifications based on ecological sources. Rather, some residents appeared to view drought tolerant or common urban species as native, suggesting a functional or cultural definition of nativity rather than a geographic or ecological definition, similar to findings in Australia (Head and Muir 2004).

Despite the evident lack of knowledge around native plants as defined ecologically, knowledge was not an important driver of native plant abundance in yards. Therefore, it is unlikely that knowledge was the limiting factor preventing native plant adoption. While other work has found that providing a combination of technical and social knowledge to residents can provoke changes in gardening behavior (van Heezik et al. 2012), we suggest that information about native species and how to garden with them may not be very important in determining native plant outcomes. Calls for increased native planting in residential spaces must go beyond simply providing information about native plants, potentially by including a normative component such as comparisons with other yards or by emphasizing native species with desirable characteristics.

Social norms have been shown to drive environmental behaviors in other contexts and are often acknowledged as important predictors of residential yard management choices (Nassauer et al. 2009, Heberlein 2012). For example, most residents surveyed in Raleigh, North Carolina were accepting of native plant landscaping, but thought their neighbors were less likely to support native landscaping (Peterson et al. 2012). Thus, social pressures may have prevented them from including as much native vegetation as they would have liked. Alternatively, in settings where native landscaping is widely accepted, social norms can enforce the use of native plants (Uren et al. 2015). Normative impacts may be particularly influential in visible front yards while back yards may be more closely guided by personal preferences, leading to measurable differences in vegetation and overall biodiversity (Larsen and Harlan 2006, Belaire et al. 2016, Ossola et al. 2019). Future work should explore the extent to which concerns about neighbor acceptance may limit adoption of native plant landscaping. Spreading awareness of high levels of social acceptance could address normative barriers and increase overall native landscaping in receptive communities (van Heezik et al. 2012).

Priorities and attitudes predict native plant abundance (H2, H3)

Respondents had positive attitudes toward native plants overall, similar to findings in New Mexico (Spinti et al. 2004). However, low reported knowledge in combination with low overall prevalence of native plants suggest that positive attitudes may be weakly held and easily changed (Heberlein 2012) due to little personal experience with native plants in a landscaping context. Increased experience with native plants could have unpredictable outcomes as people grow more familiar with them. Further investigation on this topic could explore the strength of attitudes toward native plants and their relationships to core values, norms, and identities, which can also affect attitude stability, to determine the likelihood of current positive attitudes remaining with increased experience (Heberlein 2012).

Previous research has suggested that people would plant natives if they thought they were attractive enough (Lockett et al. 2002). However, we show here that beliefs that native plants are beautiful was negatively related to yard native plant abundance. Respondents who believed native plants were beautiful also held positive attitudes toward the desert, but did not necessarily believe that native plants belonged in the city. These differences suggest an appreciation of native species in their natural habitat but a sense that they do not belong in residential yards (Head and Muir 2004). Additionally, prioritizing beautiful plants had no relationship with native plant abundance, but did predict reduced native diversity, suggesting that residents choose a subset of natives that they find most beautiful. Phoenix residential yard landscaping tends to have greater vegetation density than the natural desert (Larsen and Harlan 2006), and thus desert- and native plant-loving residents may aim to create a more natural-looking yard landscape through lower density planting, resulting in reduced native plant abundance. Another possible explanation for this surprising relationship could be that residents with more experience with native plants in their yards think they are less beautiful due to the responses of native species to urban stresses and improper care. For example, some desert shrub species become very large and sparse under high water conditions (e.g., over-irrigation), which can lead to undesirable growth forms. Regardless of the mechanism, this result shows that efforts to improve people's attitudes toward native plants are unlikely to result in greater native plant cultivation in residential landscapes.

Most previous research has shown that drought tolerance is important to residents and that native plants in arid environments are chosen to reduce irrigation needs (Potts et al. 2002, Martin et al. 2003, Kendal et al. 2012, Uren et al. 2015). Our study supports the finding that residents place importance on water needs when selecting plants. However, we found that people who identified water use as more important had fewer native plants in their front yards. One explanation for this seemingly contradictory result is the low prevalence of native plants in the Arizona horticultural flora. For example, a prominent guide to low water use landscape plants for Arizona includes only 15% native species (“Landscape plants for the Arizona desert” 2006). Native plants sold at nurseries are often unlabeled and rarely marked as drought-tolerant (Brzuszek and Harkess 2009). Given the lack of knowledge about which plants are native, residents may not be able to select native plants for drought-tolerant landscaping without marketing guidance.

The relationship between residents' reported plant selection priorities and vegetation outcomes reflects the gap between attitudes and behavior (Kollmuss and Agyeman 2002). Other research has found that concern about conservation and intentions to use less water for landscaping often don't correspond with actual landscaping decisions, but instead follow social norms and personal preferences (Larson and Brumand 2014). In this case, residents may feel that choosing low water use vegetation is important, but be unlikely to make changes to existing high water use vegetation or to select drought tolerant plants that conflict with other priorities. Our survey questions focused on plant purchasing decisions, but the observed yard vegetation is the result of accumulated plant addition, removal, and maintenance over time. Legacies of previous vegetation decisions made by developers and former residents also affect residential vegetation and therefore likely result in some disconnect between current resident priorities and yard composition (Larsen and Harlan 2006, Grove et al. 2017).

A main benefit of native plant landscaping is resource and habitat provision for native wildlife (Burghardt et al. 2009, Narango et al. 2018). While attracting wildlife was not a top priority for our respondents, we found that those who chose plants to support wildlife had more native plants, as well as more woody plants overall. Similarly, a study of residential yards in Minnesota found greater native species richness in yards where residents prioritized supporting wildlife (Cavender-Bares et al. 2020). However, Cavender-Bares et al. found no relationship between reported importance of cultivating species native to the state and native plant diversity, while we found that residents who said they prioritized natives did have more individuals, though not more species. This difference could reflect differing non-attitudinal barriers to native plant cultivation in these regions, such as horticultural availability of native species, or could show that native-focused gardeners simply plant more of the same few natives. Future research should explore drivers of native plant landscaping in a wider range of climatic and social contexts to evaluate commonalities and differences among regions.

Prioritizing low maintenance and easy-to-get plants had minimal correlations with native plant abundance. Previous research has suggested that gardeners choose native plants due in part to their low maintenance needs (Brzuszek et al. 2010), but our respondents simply had fewer plants if they prioritized low maintenance needs. Similarly, prioritizing easy to get plants was also related to overall lower yard plant abundance. Residents who prioritize spending little time and effort on their yards may be unlikely to pursue management practices such as native plant landscaping, unless they are easy.

Resident characteristics and parcel structure (H4)

Resident characteristics and parcel structure were related to native plant abundance, but not more strongly than attitudes and plant choice priorities. As expected from the luxury effect, higher income predicted increased abundance of native, succulent, and all woody plants. However, belief that native plants are beautiful and belong in the city, prioritization of natives, and prioritization of low water use plants had larger effects, at least for nativity as defined by the Arizona Sonoran Desert. Yard rock cover was also positively associated with native plant abundance, with a similar magnitude effect as income. Thus, increased xeric landscaping across the region may come with increased native plant abundance as native plants are seen as a better fit with this landscaping type. Overall woody vegetation increased with rock cover as well, suggesting that this effect may be due to an increase in woody planting (including natives) when turfgrass lawns are omitted.

Future directions for urban native plant landscaping

Our comparison of reported and observed plant nativity shows a lack of clarity among our respondents about what it means for a plant to be native. However, many respondents said that nativity is important to them when they choose a new plant. Other work has suggested that the designation of "native" is more of a value statement than a categorization related to particular functional benefits (Kendle and Rose 2000). However, consumers asked whether they thought plant designation as "native" was primarily a marketing gimmick mostly disagreed, and agreed instead that native plants provided biodiversity and air pollution benefits (Yue et al. 2012). We identify that marketing of native plants in arid environments as low water users could be one opportunity for emphasizing function-based benefits of native plants.

Enduring challenges to increasing native plant abundance in the urban environment include the selection of species to match urban needs and constraints (e.g., tolerant of urban air quality, low branch failure rates), ability of the horticultural industry to successfully propagate and distribute native species, and perceptions in the industry that such an effort will be worthwhile (Crewe 2013). Considerations of these challenges and guidance from the horticultural industry will be important in any successful campaign to increase native plant landscaping through changes in marketing or other structural fixes, as the current positive resident attitudes toward these species may be weak and subject to change with negative experiences.

An important caveat to this study is the bias in survey respondents for whom we are able to draw conclusions about native plant attitudes and cultivation. Our survey respondents included few who were non-white, renters, or lower income; thus, we cannot claim to explain how attitudes, priorities, and structural considerations affect yard landscaping among these groups. Previous work shows differences in native plant preference by ethnicity (Peterson et al. 2012) and differences in yard management priorities for homeowners and renters (Larson et al. 2009). In Phoenix, Latino and lower-income residents have more negative attitudes toward the desert, which may also impact attitudes toward native plants (Andrade et al. 2019). Further, renters, low income residents, and ethnic minorities may be more likely to reside in historically disadvantaged neighborhoods with less vegetation and reduced access to boutique horticultural sources that sell more native plants (Grove et al. 2017, Avolio et al. 2018). Much work in residential landscapes has focused primarily on higher income white homeowners, and thus, we suggest that it is particularly important for future work to systematically include historically understudied groups in order to develop a more inclusive and representative understanding of residential landscapes.

CONCLUSIONS

Native plants can provide important resources for urban native wildlife, but currently make up less than a third of front yard residential woody vegetation. Phoenix residents who responded to our survey reported positive attitudes toward native plants, but lacked knowledge of their identification and care. In this sample, priorities for native plants, belief that native plants belong in the city, household income, and yard rock cover all positively relate to yard native plant abundance while belief that native plants are beautiful and prioritizing low water use plants are negatively related to native abundance. Thus, both attitudinal and structural factors (but not knowledge) drive native plant landscaping in this arid residential context. While one step toward integrating native plants into the urban landscape is increased education about their identification, value, and care, our results suggest that this approach is unlikely to change native plant adoption. Rather, structural barriers, such as native plant availability and cost, and opportunities, such as the current lack of labeling and marketing of drought-tolerant natives, should be addressed first. Positive attitudes should be monitored to ensure that increased experience with native plants does not change weakly held opinions, but attitudes are currently supportive of native plants in the urban landscape. However, further research is needed on how attitudes and native plant cultivation vary among residents of different socioeconomic status and ethnicity before applying generalized interventions. Where native plants are identified as uniquely valuable for wildlife, strategies can work with existing attitudes to support greater inclusion of native plants in the residential landscape.

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REFERENCES

Andrade, R., K. L. Larson, D. M. Hondula, and J. Franklin. 2019. Social–spatial analyses of attitudes toward the desert in a southwestern U.S. city. Annals of the American Association of Geographers 109:1845–1864.

Aronson, M. F. J., C. A. Lepczyk, K. L. Evans, M. A. Goddard, S. B. Lerman, J. S. MacIvor, C. H. Nilon, and T. Vargo. 2017. Biodiversity in the city: key challenges for urban green space management. Frontiers in Ecology and the Environment 15:189–196.

Avolio, M. L., D. E. Pataki, T. L. E. Trammell, and J. Endter-Wada. 2018. Biodiverse cities: the nursery industry, homeowners, and neighborhood differences drive urban tree composition. Ecological Monographs 88:259–276.

Bailey, L. H., and E. Z. Bailey. 1976. Hortus third: a concise dictionary of plants cultivated in the United States and Canada. Macmillan Publishing Company, New York, New York, USA.

Behrendt, S. 2014. lm.beta: add standardized regression coefficients to lm-objects. R package version 1.5-1. https://cran.r-project.org/package=lm.beta

Belaire, J. A., L. M. Westphal, and E. S. Minor. 2016. Different social drivers, including perceptions of urban wildlife, explain the ecological resources in residential landscapes. Landscape Ecology 31:401–413.

Bigsby, K. M., M. R. McHale, and G. R. Hess. 2014. Urban morphology drives the homogenization of tree cover in Baltimore, MD, and Raleigh, NC. Ecosystems 17:212–227.

Brzuszek, R. F., and R. L. Harkess. 2009. Green industry survey of native plant marketing in the southeastern United States. HortTechnology 19:168–172.

Brzuszek, R. F., R. L. Harkess, and L. Kelly. 2010. Survey of master gardener use of native plants in the southeastern United States. HortTechnology 20:462–466.

Brzuszek, R. F., R. L. Harkess, and S. J. Mulley. 2007. Landscape architects' use of native plants in the southeastern United States. HortTechnology 17:78–81.

Burghardt, K. T., D. W. Tallamy, and W. G. Shriver. 2009. Impact of native plants on bird and butterfly biodiversity in suburban landscapes. Conservation Biology 23:219–224.

Cavender-Bares, J., J. Padullés Cubino, W. D. Pearse, S. E. Hobbie, A. J. Lange, S. Knapp, and K. C. Nelson. 2020. Horticultural availability and homeowner preferences drive plant diversity and composition in urban yards. Ecological Applications 30:e02082.

Cook, E. M., S. J. Hall, and K. L. Larson. 2012. Residential landscapes as social-ecological systems: a synthesis of multi-scalar interactions between people and their home environment. Urban Ecosystems 15:19–52.

Crewe, K. 2013. Arizona native plants and the urban challenge. Landscape Journal 32:215–229.

Derby Lewis, A., M. J. Bouman, A. M. Winter, E. A. Hasle, D. F. Stotz, M. K. Johnston, K. R. Klinger, A. Rosenthal, and C. A. Czarnecki. 2019. Does nature need cities? Pollinators reveal a role for cities in wildlife conservation. Frontiers in Ecology and Evolution 7:220.

Fischer, A., B. Bednar-Friedl, F. Langers, M. Dobrovodská, N. Geamana, K. Skogen, and M. Dumortier. 2011. Universal criteria for species conservation priorities? Findings from a survey of public views across Europe. Biological Conservation 144:998–1007.

Frick, J., F. G. Kaiser, and M. Wilson. 2004. Environmental knowledge and conservation behavior: exploring prevalence and structure in a representative sample. Personality and Individual Differences 37:1597–1613.

Frost, D. 2016, January. An Eye on Every Drop. American Planning Association Magazine.

Gerrish, E., and S. L. Watkins. 2018. The relationship between urban forests and income: a meta-analysis. Landscape and Urban Planning 170:293–308.

Goddard, M. A., A. J. Dougill, and T. G. Benton. 2009. Scaling up from gardens: biodiversity conservation in urban environments. Trends in Ecology & Evolution 25:90–98.

Goddard, M. A., A. J. Dougill, and T. G. Benton. 2013. Why garden for wildlife? Social and ecological drivers, motivations and barriers for biodiversity management in residential landscapes. Ecological Economics 86:258–273.

Groffman, P. M., J. Cavender-Bares, N. D. Bettez, J. M. Grove, S. J. Hall, J. B. Heffernan, S. E. Hobbie, K. L. Larson, J. L. Morse, C. Neill, K. Nelson, J. O'Neil-Dunne, L. Ogden, D. E. Pataki, C. Polsky, R. Roy Chowdhury, and M. K. Steele. 2014. Ecological homogenization of urban USA. Frontiers in Ecology and the Environment 12:74–81.

Grove, M., L. Ogden, S. Pickett, C. Boone, G. Buckley, D. H. Locke, C. Lord, and B. Hall. 2017. The legacy effect: understanding how segregation and environmental injustice unfold over time in Baltimore. Annals for the American Association of Geographers 108:524–537.

Head, L., and P. Muir. 2004. Nativeness, invasiveness, and nation in Australian plants. The Geographical Review 94:199–217.

Head, L., and P. Muir. 2006. Suburban life and the boundaries of nature: resilience and rupture in Australian backyard gardens. Transactions of the Institute of British Geographers 31:505–524.

Heavenrich, H., and S. J. Hall. 2016. Elevated soil nitrogen pools after conversion of turfgrass to water-efficient residential landscapes. Environmental Research Letters 11:084007.

Heberlein, T. A. 2012. Navigating environmental attitudes. Oxford University Press, New York, New York, USA.

Hooper, V. H., J. Endter-Wada, and C. W. Johnson. 2008. Theory and practice related to native plants: a case study of Utah landscape professionals. Landscape Journal 27:127–141.

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

Hoyle, H., J. Hitchmough, and A. Jorgensen. 2017. Attractive, climate-adapted and sustainable? Public perception of non-native planting in the designed urban landscape. Landscape and Urban Planning 164:49–63.

Kartesz, J. T. 2015. North American plant atlas. The Biota of North America Program (BONAP), Chapel Hill, North Carolina, USA. http://bonap.net/napa.

Kauth, P. J., and H. E. Pérez. 2011. Industry survey of the native wildflower market in Florida. HortTechnology 21:779–788.

Kendal, D., K. J. H. Williams, and N. S. G. Williams. 2012. Plant traits link people's plant preferences to the composition of their gardens. Landscape and Urban Planning 105:34–42.

Kendle, A. D., and J. E. Rose. 2000. The aliens have landed! What are the justifications for "native only" policies in landscape plantings? Landscape and Urban Planning 47:19–31.

Kim, M., and J. E. Hunter. 1993. Attitude-behavior relations: a meta-analysis of attitudinal relevance and topic. Journal of Communication 43:101–142.

Kollmuss, A., and J. Agyeman. 2002. Mind the gap: why do people behave environmentally and what are the barriers to pro-environmental behavior? Environmental Education Research 8:239–260.

Landscape plants for the Arizona desert. 2006. Arizona Municipal Water Users Association (AMWUA). http://www.amwua.org/plants.

Larsen, L., and S. L. Harlan. 2006. Desert dreamscapes: residential landscape preference and behavior. Landscape and Urban Planning 78:85–100.

Larson, K. L. 2010. An integrated theoretical approach to understanding the sociocultural basis of multidimensional environmental attitudes. Society & Natural Resources 23:898–907.

Larson, K. L., R. Andrade, A. York, D. Childers, P. Coseo, S. Earl, M. Ehlenz, N. Grimm, S. J. Hall, S. Harlan, D. Hondula, S. B. Lerman, D. Pfeiffer, J. Ripplinger, K. Smith, P. Warren, M. Watkins, D. White, S. Wittlinger, M. Wright, and A. Wutich. 2017a. The Phoenix Area Social Survey IV: Linking Social and Biophysical Dynamics in Urban Neighborhoods. A report from the Central Arizona–Phoenix Long-Term Ecological Research (CAP LTER) project. Tempe, Arizona.

Larson, K. L., and J. Brumand. 2014. Paradoxes in landscape management and water conservation: examining neighborhood norms and institutional forces. Cities and the Environment 7:6.

Larson, K. L., D. Casagrande, S. L. Harlan, and S. T. Yabiku. 2009. Residents' yard choices and rationales in a desert city: social priorities, ecological impacts, and decision tradeoffs. Environmental Management 44:921–937.

Larson, K. L., E. Cook, C. Strawhacker, and S. J. Hall. 2010. The influence of diverse values, ecological structure, and geographic context on residents' multifaceted landscaping decisions. Human Ecology 38:747–761.

Larson, K. L., K. C. Nelson, S. R. Samples, S. J. Hall, N. Bettez, J. Cavender-Bares, P. M. Groffman, M. Grove, J. B. Heffernan, S. E. Hobbie, J. Learned, J. L. Morse, C. Neill, L. A. Ogden, J. O'Neil-Dunne, D. E. Pataki, C. Polsky, R. Roy Chowdhury, M. Steele, and T. L. E. Trammell. 2016. Ecosystem services in managing residential landscapes: priorities, value dimensions, and cross-regional patterns. Urban Ecosystems 19:95–113.

Levine, D. S., and M. J. Strube. 2012. Environmental attitudes, knowledge, intentions and behaviors among college students. Journal of Social Psychology 152:308–326.

Lockett, L., T. Montague, C. McKenney, and D. Auld. 2002. Assessing public opinion on water conservation and water conserving landscapes in the semiarid southwestern United States. HortTechnology 12:392–396.

Martin, C. A., K. A. Peterson, and L. B. Stabler. 2003. Residential landscaping in Phoenix, Arizona, U.S.: practices and preferences relative to covenants, codes, and restrictions. Journal of Arboriculture 29:9–17.

Martini, N. F., and K. C. Nelson. 2015. The role of knowledge in residential lawn management. Urban Ecosystems 18:1031–1047.

McKinney, M. L. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127:247–260.

Narango, D. L., D. W. Tallamy, and P. P. Marra. 2018. Nonnative plants reduce population growth of an insectivorous bird. Proceedings of the National Academy of Sciences USA 115:11549–11554.

Nassauer, J. I., Z. Wang, and E. Dayrell. 2009. What will the neighbors think? Cultural norms and ecological design. Landscape and Urban Planning 92:282–292.

Ossola, A., D. Locke, B. Lin, and E. Minor. 2019. Greening in style: urban form, architecture and the structure of front and backyard vegetation. Landscape and Urban Planning 185:141–157.

Padullés Cubino, J., M. L. Avolio, M. M. Wheeler, K. L. Larson, S. E. Hobbie, J. Cavender-Bares, S. J. Hall, K. C. Nelson, T. L. E. Trammell, C. Neill, D. E. Pataki, J. M. Grove, and P. M. Groffman. 2020. Linking yard plant diversity to homeowners' landscaping priorities across the U.S. Landscape and Urban Planning 196:103730.

Padullés Cubino, J., J. Cavender-Bares, S. E. Hobbie, D. E. Pataki, M. L. Avolio, L. E. Darling, K. L. Larson, S. J. Hall, P. M. Groffman, T. L. E. Trammell, M. K. Steele, J. M. Grove, and C. Neill. 2018. Drivers of plant species richness and phylogenetic composition in urban yards at the continental scale. Landscape Ecology 34:63–77.

Pardee, G. L., and S. M. Philpott. 2014. Native plants are the bee's knees: local and landscape predictors of bee richness and abundance in backyard gardens. Urban Ecosystems 17:641–659.

Peterson, M. N., B. Thurmond, M. McHale, S. Rodriguez, H. D. Bondell, and M. Cook. 2012. Predicting native plant landscaping preferences in urban areas. Sustainable Cities and Society 5:70–76.

Potts, L. E., M. J. Roll, and S. J. Wallner. 2002. Colorado native plant survey — voices of the green industry. Native Plants Journal 3:121–125.

R Core Team. 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ripley, B., B. Venables, D. M. Bates, K. Hornik, A. Gebhardt, and D. Firth. 2019. Support Functions and Datasets for Venables and Ripley's MASS. R package version 7.3-51.4. https://CRAN.R-project.org/package=MASS

Roman, L. A., J. J. Battles, and J. R. McBride. 2014. Determinants of establishment survival for residential trees in Sacramento County, CA. Landscape and Urban Planning 129:22–31.

Schultz, P. W. 2011. Conservation means behavior. Conservation Biology 25:1080–1083.

Spinti, J. E., R. St. Hilaire, and D. M. VanLeeuwen. 2004. Balancing landscape preferences and water conservation in a desert community. HortTechnology 14:72–77.

St. Hilaire, R., D. M. VanLeeuwen, and P. Torres. 2010. Landscape preferences and water conservation choices of residents in a high desert environment. HortTechnology 20:308–314.

Threlfall, C. G., L. Mata, J. A. Mackie, A. K. Hahs, N. E. Stork, N. S. G. Williams, and S. J. Livesley. 2017. Increasing biodiversity in urban green spaces through simple vegetation interventions. Journal of Applied Ecology 54:1874–1883.

U.S. Census Bureau. 2019b. Highlights of annual 2019 characteristics of new housing. https://www.census.gov/construction/chars/highlights.html.

Uren, H. V., P. L. Dzidic, and B. J. Bishop. 2015. Exploring social and cultural norms to promote ecologically sensitive residential garden design. Landscape and Urban Planning 137:76–84.

USDA NRCS. 2020. The PLANTS database. National Plant Data Team, Greensboro, North Carolina, USA. http://plants.usda.gov.

van Heezik, Y. M., K. J. M. Dickinson, and C. Freeman. 2012. Closing the gap: communicating to change gardening practices in support of native biodiversity in urban private gardens. Ecology and Society 17:34.

Villaseñor, J. L. 2016. Checklist of the native vascular plants of Mexico. Revista Mexicana de Biodiversidad 87:559–902.

Wheeler, M. M., K. L. Larson, and R. Andrade. 2020. Attitudinal and structural drivers of preferred versus actual residential landscapes in a desert city. Urban Ecosystems 23:659–673.

White, A., J. B. Fant, K. Havens, M. Skinner, and A. T. Kramer. 2018. Restoring species diversity: assessing capacity in the U.S. native plant industry. Restoration Ecology 26:605–611.

Yue, C., T. Hurley, and N. O. Anderson. 2012. Heterogeneous consumer preferences for native and invasive plants: evidence from experimental auctions. HortScience 47:1091–1095.

Zube, E. H., D. E. Simcox, and C. S. Law. 1986. The oasis image in two desert cities. Landscape Research 11:7–11.

TABLE 1. Neighborhood characteristics and sampling effort.

Neighborhood	Dominant landscape type	Home age (mean years ± std. dev.)[1]	Household median income ($)[2]	Plant surveys conducted	Resident survey responses	Response rate (%)
Old Hispanic Core	Mesic	65 ± 7	35,000	105	6	5.9
Historic Palms District	Mesic	80 ± 7	89,000	95	39	40.6
New Xeric Tracts	Xeric	24 ± 5	63,000	107	29[3]	27.4
Wealthy Mountain Oasis	Xeric	24 ± 2	150,000	109	30	28.6
Unknown[4]					1	
Total				416	105	25.7

[1] From tax assessor reported year built, average age in 2018 for all parcels with plant surveys

[2] As reported for most closely overlapping 2017 U.S. Census tract

[3] Includes one respondent for whom vegetation data were not collected

[4] Unique identifier removed from completed survey

TABLE 2. Summary statistics for respondent native plant knowledge, attitudes, priorities, and characteristics, and parcel structure. The desert attitudinal scale was calculated with negative items reversed such that higher values indicate more positive attitudes. Total respondents = 105.

	Mean ± sd	Median	Range	N
Knowledge[1]				
Native plant knowledge scale (corr = 0.62)	2.9 ± 1.1	3	1 - 5	104
Know about gardening with	2.7 ± 1.2	3	1 - 5	105
Know how to determine nativity	3.0 ± 1.2	3	1 - 5	104
Attitudes[1]				
Native plants do not belong in city	1.5 ± 0.9	1	1 - 5	105
Native plants are beautiful	4.3 ± 0.9	5	1 - 5	105
Desert scale (alpha = 0.89)	4.2 ± 0.9	4.5	1.5 - 5	104
Wasteland[2]	1.3 ± 0.7	1	1 - 4	104
Very special	3.9 ± 1.3	4	1 - 5	104
Beautiful	4.3 ± 1.0	5	1 - 5	104
Nice place to spend time	4.0 ± 1.2	4	1 - 5	104
Plant Choice Priorities[3]				
Low water use	3.3 ± 0.7	3	2 - 4	103
Beautiful	3.2 ± 0.6	3	2 - 4	102
Low maintenance	3.0 ± 0.8	3	1 - 4	103
Native	2.9 ± 0.9	3	1 - 4	102
Has spines	2.6 ± 1.0	3	1 - 4	103
Attracts wildlife	2.6 ± 0.9	3	1 - 4	103
Easy to get	2.5 ± 0.9	3	1 - 4	103
Resident Characteristics				
Percent of life in Phoenix	49 ± 29	45	0 - 100	101
Household income (ordinal)	7.5 ± 3.3	7.5	1 - 11	84
Education (ordinal)	4 ± 1	4	1 - 5	102
Parcel Structure				
Front yard area (m^2)	221 ± 155	185	62 - 1,165	104
Front yard rock cover (%)	27 ± 23	33	0 - 69	104

[1] Levels: strongly disagree (1), neither agree nor disagree (3), strongly agree (5)

[2] Negatively worded item, reversed in combined scale

[3] Levels: not at all important (1) to very important (4)

TABLE 3. Predictors of woody plant, succulent, and native plant abundance in 80 Phoenix front yards. Predictors that are not included in the model for each response variable are left blank.

	Native to AZ Sonoran		Native to Sonoran		Succulents		All woody plants	
	Std Beta	P value	Std Beta	P value	Std Beta	P value	Std Beta	P value
Knowledge								
Knowledge of native plants	-	-	-	-	-	-	-0.004	0.01
Attitudes								
Native plants don't belong in the city	-0.06	0.0002	-0.02	0.003	-0.01	0.05	-	-
Native plants are beautiful	-0.07	<0.0001	-0.06	<0.0001	-0.08	<0.0001	-0.016	<0.0001
Attitudes toward the desert	-	-	0.03	0.0001	0.02	0.003	-	-
Plant Choice Priorities								
Low water use	-0.06	<0.0001	-0.03	<0.0001	-	-	0.004	0.02
Beautiful	-	-	-	-	0.01	0.01	0.009	<0.0001
Low maintenance	0.02	0.13	-	-	-0.02	0.002	-0.006	0.0005
Native	0.07	<0.0001	0.03	<0.0001	-	-	-	-
Has spines	0.03	0.008	0.01	0.03	0.02	0.007	0.005	0.002
Attracts wildlife	0.04	0.001	0.02	0.01	0.05	<0.0001	0.010	<0.0001
Easy to get	-0.02	0.08	-	-	-0.01	0.03	-0.003	0.10
Resident Characteristics								
Percent of life in Phoenix	-	-	-	-	-	-	0.008	<0.0001
Household income	0.06	<0.0001	0.05	<0.0001	0.06	<0.0001	0.023	<0.0001
Education	0.02	0.07	-	-	-	-	-	
Parcel Structure								
Front yard area	-	-	-	-	-0.02	0.003	0.006	<0.0001
Front yard rock cover	0.06	<0.0001	0.05	<0.0001	0.06	<0.0001	0.005	0.01
Model pseudo R squared	0.489		0.413		0.439		0.542	

TABLE 4. Predictors of Arizona Sonoran Desert native plant species richness in 80 Phoenix front yards. Predictors that are not included in the model for each response variable are left blank.

	Std beta	P value
Knowledge		
Knowledge of native plants	-	-
Attitudes		
Native plants don't belong in the city	-0.11	0.03
Native plants are beautiful	-0.23	0.000004
Attitudes toward the desert	0.08	0.10
Plant Choice Priorities		
Low water use	-0.12	0.01
Beautiful	-0.10	0.04
Low maintenance	-	-
Native	-	-
Has spines	-	-
Attracts wildlife	0.13	0.01
Easy to get	-	-
Resident Characteristics		
Percent of life in Phoenix	-	-
Household income	0.09	0.08
Education	-	-
Parcel Structure		
Front yard area	-	-
Front yard rock cover	0.16	0.002

Notes: Model pseudo $R^2 = 0.368$

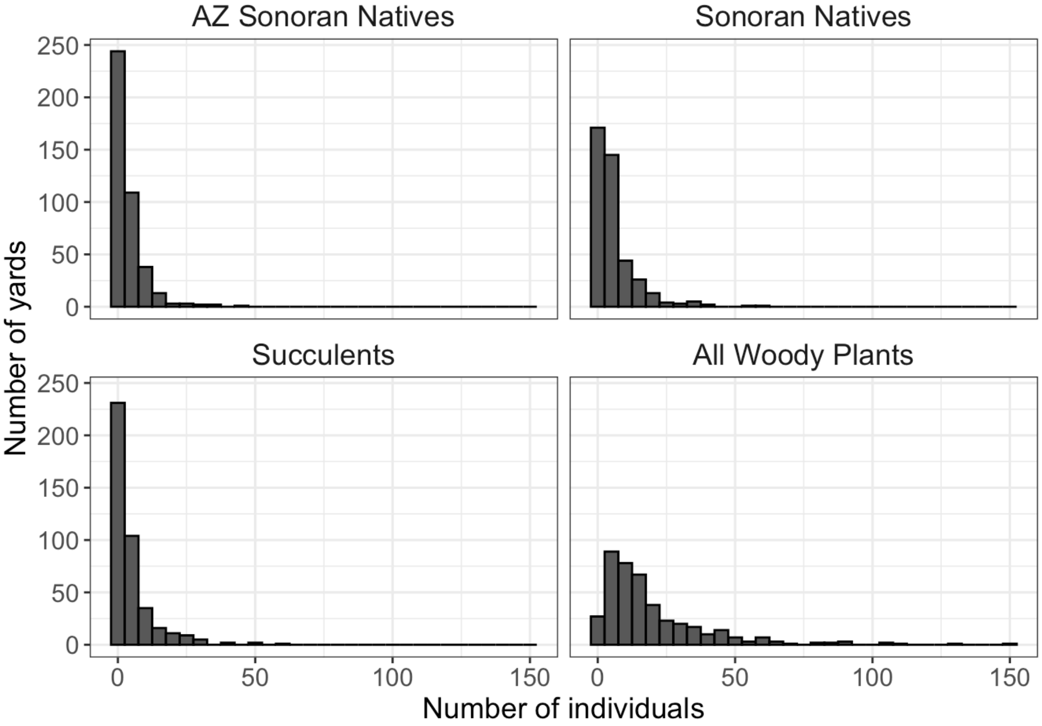

FIG. 1. Plant abundance in 416 surveyed yards.

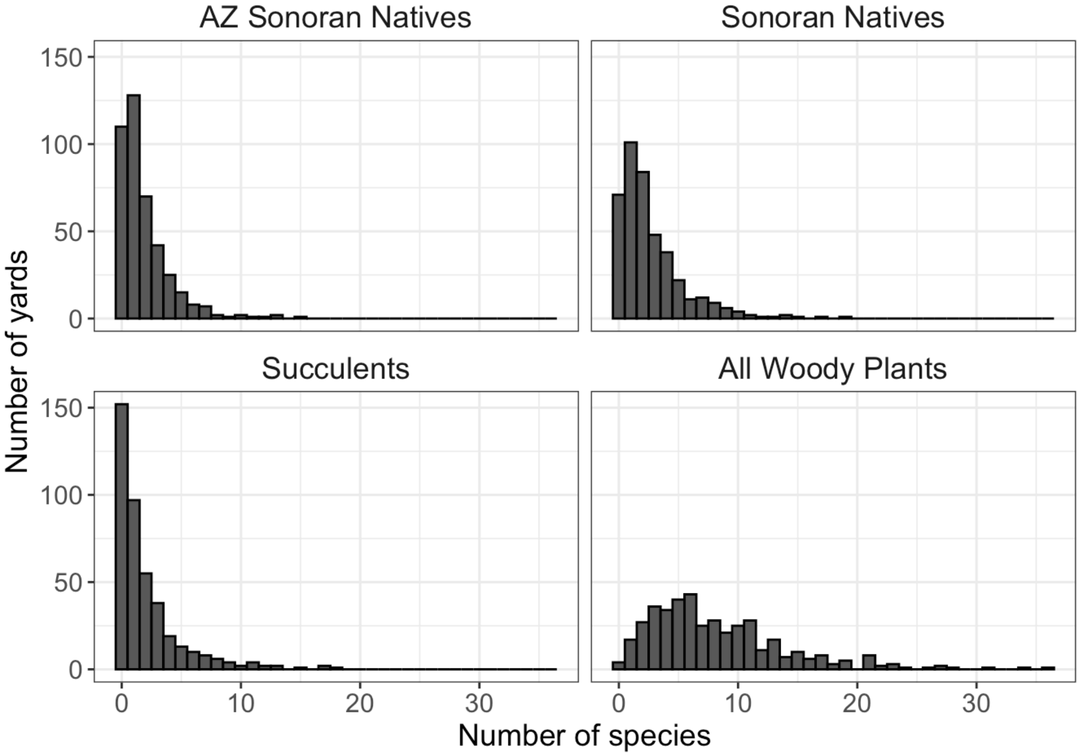

FIG. 2. Species richness in 416 surveyed yards.

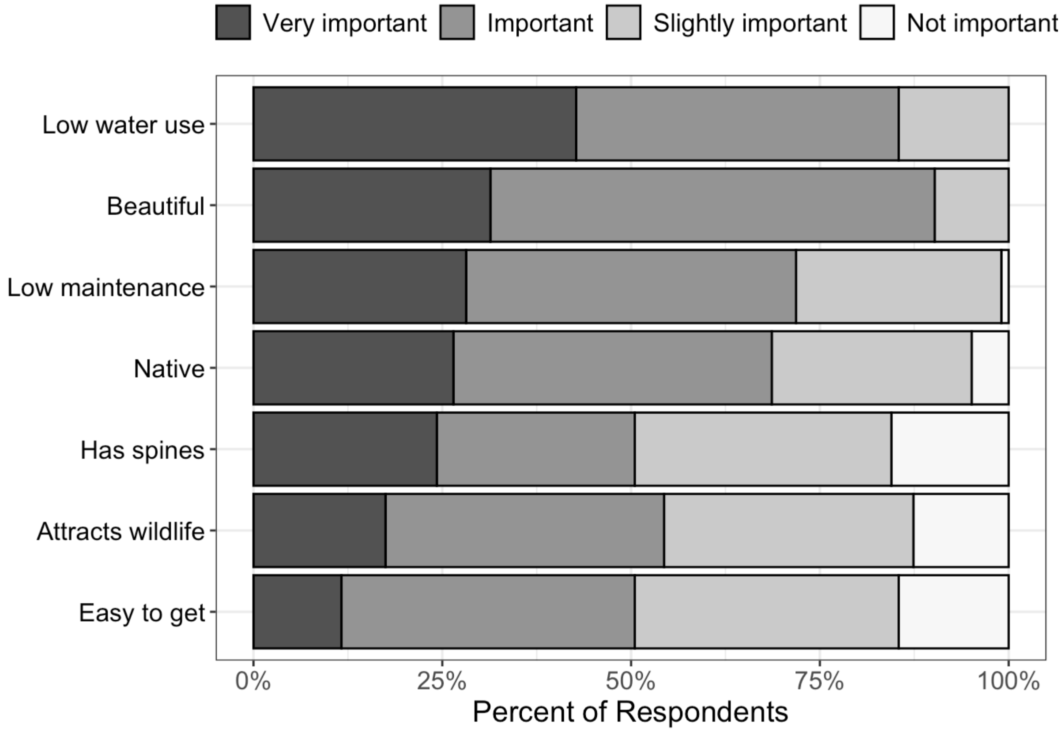

FIG. 3. Reported importance of each priority for respondents when choosing a new plant for their yard. N = 103.

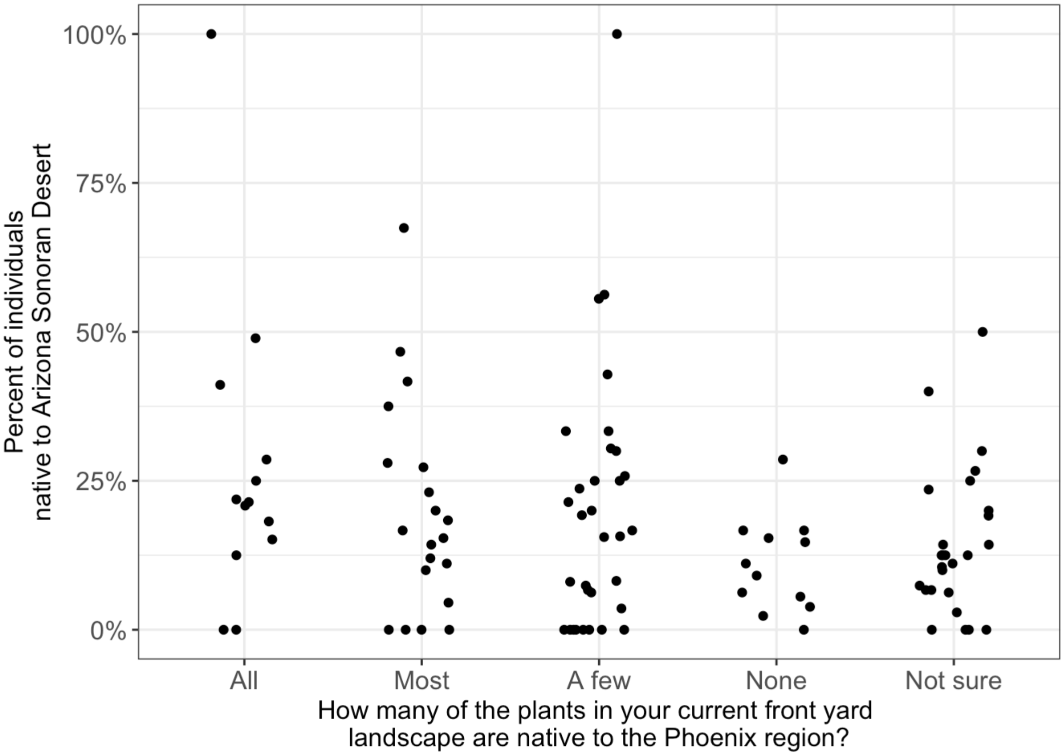

FIG. 4. Respondent classifications of the number of native plants in their front yard (x-axis) compared to vegetation survey results (y-axis). Each point represents one respondent/yard. Points are horizontally offset for clarity.

CHAPTER 6
CONCLUDING REMARKS

Through this dissertation research, I have explored drivers of biodiversity change over time in urban and urban-influenced environments, revealing both social and environmental predictors of ecological outcomes. In the following sections, I highlight the major findings from each chapter and identify implications and future research directions.

MAJOR RESEARCH FINDINGS

Chapter 2: The theory of urban ecological convergence through biotic homogenization and other mechanisms has been widely studied using a variety of methods. However, the diversity of approaches to testing urban homogenization have resulted in a disparate body of literature with mixed findings. Most studies found taxonomic homogenization of plants and birds. Variability in defining homogenization and in comparisons used to test for homogenization led to uncertainty in other findings. This synthesis identified a need for future mechanistic research on urban homogenization, testing proposed mechanisms using the appropriate comparisons, spatial scales, taxa, and metrics. Improved mechanistic understanding of patterns of urban biodiversity change will in turn help identify conservation needs and opportunities, in addition to building robust urban ecological theory.

Chapter 3: Nitrogen deposition is considered one of the greatest threats to global biodiversity, but the effects of nitrogen enrichment in arid environments is greatly understudied (Bobbink et al. 2010). This long-term experimental study provided causal evidence of reduced desert annual plant biodiversity with nitrogen enrichment, regardless of water availability. Further, urban preserves hosted lower annual plant diversity than preserves outside of the city. The loss of charismatic wildflower species in urban preserves may be particularly damaging both for dependent wildlife and for urban residents.

Chapter 4: Urban landscapes are often treated as static over time (La Sorte et al. 2014b, Roman et al. 2018). I show that residential yard woody plant communities experienced high compositional turnover in a ten-year period. Residents reported making changes to their landscapes, suggesting that most residents engage in active management that alters the plant community. Thus, there are opportunities to engage residents in choosing changes that will benefit biodiversity, such as increasing native plantings. This work contributes to fundamental understanding of plant community dynamics in residential spaces, which have rarely been studied over time. Improved understanding of temporal dynamics and drivers of change in urban landscapes is important for predicting and modeling future urban scenarios (Raudsepp-Hearne et al. 2019, Iwaniec et al. 2020).

Chapter 5: Native plants can provide unique resources to native wildlife in urban areas (Burghardt et al. 2009, Narango et al. 2018). However, less than a third of plants in Phoenix residential front yards were native. Though most residents were uncertain of native plant care and identification, they had largely positive attitudes toward native plants. More native plants were found in yards where residents prioritized native plants and habitat provision for wildlife and where residents believed native plants belong in the city. However, residents who wanted low water use plants cultivated fewer native individuals, despite their adaptations to the desert environment. Consistent with social science theory, we observed a disconnect between knowledge and behavior, suggesting that increasing technical knowledge is unlikely to increase native plant abundance in residential spaces (Kollmuss and Agyeman 2002). Given the high rates of change observed in yard communities, there may be opportunities to remove structural barriers, such as lack of native plant availability, high price, and normative constraints, to encourage greater adoption of native plants among receptive residents.

IMPLICATIONS AND FUTURE RESEARCH

Application of interdisciplinary urban ecological research in urban design and planning practices is a growing frontier, with clear possibilities for improving both human and ecological health in cities (Taylor and Hochuli 2015, Mata et al. 2020). In this dissertation, I have advanced the mechanistic understanding of social and ecological drivers of urban environmental outcomes, including the effects of N enrichment on near-urban environments and the relationship between resident attitudes and yard landscaping. These results can be applied to inform preserve management action, to promote ecologically beneficial plant choice in residential yards, and to identify locations and groups of people who may be most amenable to making changes to support urban conservation goals.

Further work is needed to provide more broadly generalizable urban conservation guidance. As was revealed by the literature review in Chapter 2, context and approach can have important impacts on the conclusions reached by urban ecological research. My research was primarily conducted in a single desert city in the United States. Thus, additional work is necessary to understand whether, for example, tropical and temperate cities experience the same rate of vegetation turnover. While the integration of social and ecological work has greatly furthered urban ecology, representative social sampling as well as biophysical sampling is an important challenge. Ethnic minorities, socioeconomically disadvantaged populations, small cities, and non-Anglo countries are poorly represented in the urban social-ecological literature, and future research should aim to better include these groups.

Urban ecology has great potential to improve the lives of people and contribute to solutions to the ongoing global biodiversity crisis. Focus on research applications and co-development of research products with end users can help increase the impact of academic work on urban planning and management. Through this research and in the future, I aim to develop social-ecological research that expands our understanding of temporal change in urban environments to guide urban conservation action into the future.

REFERENCES

Bobbink, R., K. Hicks, J. Galloway, T. Spranger, R. Alkemade, M. Ashmore, M. Bustamante, S. Cinderby, E. Davidson, F. Dentener, B. Emmett, J.-W. Erisman, M. Fenn, F. Gilliam, A. Nordin, L. Pardo, and W. De Vries. 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. Ecological Applications 20:30–59.

Burghardt, K. T., D. W. Tallamy, and W. G. Shriver. 2009. Impact of native plants on bird and butterfly biodiversity in suburban landscapes. Conservation Biology 23:219–224.

Iwaniec, D. M., E. M. Cook, M. J. Davidson, M. Berbés-Blázquez, and N. B. Grimm. 2020. Integrating existing climate adaptation planning into future visions: a strategic scenario for the central Arizona–Phoenix region. Landscape and Urban Planning 200:103820.

Kollmuss, A., and J. Agyeman. 2002. Mind the gap: why do people behave environmentally and what are the barriers to pro-environmental behavior? Environmental Education Research 8:239–260.

La Sorte, F. A., M. W. Tingley, and A. H. Hurlbert. 2014. The role of urban and agricultural areas during avian migration: an assessment of within-year temporal turnover. Global Ecology and Biogeography 23:1225–1234.

Mata, L., C. E. Ramalho, J. Kennedy, K. M. Parris, L. Valentine, M. Miller, S. Bekessy, S. Hurley, and Z. Cumpston. 2020. Bringing nature back into cities. People and Nature 2:350–368.

Narango, D. L., D. W. Tallamy, and P. P. Marra. 2018. Nonnative plants reduce population growth of an insectivorous bird. Proceedings of the National Academy of Sciences USA 115:11549–11554.

Raudsepp-Hearne, C., G. D. Peterson, E. M. Bennett, R. Biggs, A. V. Norström, L. Pereira, J. Vervoort, D. M. Iwaniec, T. McPhearson, P. Olsson, T. Hichert, M. Falardeau, and A. Jiménez Aceituno. 2019. Seeds of good anthropocenes: developing sustainability scenarios for Northern Europe. Sustainability Science 15:605–617.

Roman, L. A., H. Pearsall, T. S. Eisenman, T. M. Conway, R. T. Fahey, S. Landry, J. Vogt, N. S. van Doorn, J. M. Grove, D. H. Locke, A. C. Bardekjian, J. J. Battles, M. L. Cadenasso, C. C. K. van den Bosch, M. Avolio, A. Berland, G. D. Jenerette, S. K. Mincey, D. E. Pataki, and C. Staudhammer. 2018. Human and biophysical legacies shape contemporary urban forests: a literature synthesis. Urban Forestry & Urban Greening 31:157–168.

Taylor, L., and D. F. Hochuli. 2015. Creating better cities: how biodiversity and ecosystem functioning enhance urban residents' wellbeing. Urban Ecosystems 18:747–762.

REFERENCES

Ackerman, D., D. B. Millet, and X. Chen. 2019. Global estimates of inorganic nitrogen deposition across four decades. Global Biogeochemical Cycles 33:100–107.

Adler, F. R., and C. J. Tanner. 2013. Urban ecosystems: ecological principles for the built environment. Cambridge University Press, New York, New York, USA.

Adondakis, S., and D. L. Venable. 2004. Dormancy and germination in a guild of Sonoran Desert annuals. Ecology 85:2582–2590.

Aguilera, L. E., J. R. Gutiérrez, and P. L. Meserve. 1999. Variation in soil micro-organisms and nutrients underneath and outside the canopy of *Adesmia bedwellii* (Papilionaceae) shrubs in arid coastal Chile following drought and above average rainfall. Journal of Arid Environments 42:61–70.

Alvey, A. A. 2006. Promoting and preserving biodiversity in the urban forest. Urban Forestry & Urban Greening 5:195–201.

Anderson, M. J., T. O. Crist, J. M. Chase, M. Vellend, B. D. Inouye, A. L. Freestone, N. J. Sanders, H. V. Cornell, L. S. Comita, K. F. Davies, S. P. Harrison, N. J. B. Kraft, J. C. Stegen, and N. G. Swenson. 2011. Navigating the multiple meanings of β diversity: a roadmap for the practicing ecologist. Ecology Letters 14:19–28.

Andrade, R., K. L. Larson, D. M. Hondula, and J. Franklin. 2019. Social–spatial analyses of attitudes toward the desert in a southwestern U.S. city. Annals of the American Association of Geographers 109:1845–1864.

Angert, A. L., J. L. Horst, T. E. Huxman, and D. L. Venable. 2010. Phenotypic plasticity and precipitation response in Sonoran Desert winter annuals. American Journal of Botany 97:405–411.

Angert, A. L., T. E. Huxman, G. A. Barron-Gafford, K. L. Gerst, and D. L. Venable. 2007. Linking growth strategies to long-term population dynamics in a guild of desert annuals. Journal of Ecology 95:321–331.

Angert, A. L., T. E. Huxman, P. Chesson, and D. L. Venable. 2009. Functional tradeoffs determine species coexistence via the storage effect. Proceedings of the National Academy of Sciences USA 106:11641–11645.

Aronson, M. F. J., C. A. Lepczyk, K. L. Evans, M. A. Goddard, S. B. Lerman, J. S. MacIvor, C. H. Nilon, and T. Vargo. 2017. Biodiversity in the city: key challenges for urban green space management. Frontiers in Ecology and the Environment 15:189–196.

Aronson, M. F. J., F. A. La Sorte, C. H. Nilon, M. Katti, M. A. Goddard, C. A. Lepczyk, P. S. Warren, N. S. G. Williams, S. Cilliers, B. Clarkson, C. Dobbs, R. Dolan, M. Hedblom, S. Klotz, J. L. Kooijmans, I. Kühn, I. MacGregor-Fors, M. McDonnell, U. Mörtberg, P. Pyšek, S. Siebert, J. Sushinsky, P. Werner, and M. Winter. 2014. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. Proceedings of the Royal Society B: Biological Sciences 281:20133330.

Asmus, U., and G. L. Rapson. 2014. Floristic homogeneity underlies environmental diversification of northern New Zealand urban areas. New Zealand Journal of Botany 52:285–303.

Avolio, M. L., D. E. Pataki, S. Pincetl, T. W. Gillespie, G. D. Jenerette, and H. R. McCarthy. 2015. Understanding preferences for tree attributes: the relative effects of socio-economic and local environmental factors. Urban Ecosystems 18:73–86.

Avolio, M. L., D. E. Pataki, T. L. E. Trammell, and J. Endter-Wada. 2018. Biodiverse cities: the nursery industry, homeowners, and neighborhood differences drive urban tree composition. Ecological Monographs 88:259–276.

Bailey, L. H., and E. Z. Bailey. 1976. Hortus third: a concise dictionary of plants cultivated in the United States and Canada. Macmillan Publishing Company, New York, New York, USA.

Bartoń, K. 2018. MuMIn: multi-model inference. R package version 1.43.6. https://CRAN.R-project.org/package=MuMIn

Baselga, A. 2010. Partitioning the turnover and nestedness components of beta diversity. Global Ecology and Biogeography 19:134–143.

Bates, D., M. Mächler, B. M. Bolker, and S. C. Walker. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67:1–48.

Beguería, S., and S. M. Vicente-Serrano. 2017. SPEI: calculation of the standardised precipitation-evapotranspiration index. R package version 1.7. https://CRAN.R-project.org/package=SPEI

Behrendt, S. 2014. lm.beta: add standardized regression coefficients to lm-objects. R package version 1.5-1. https://cran.r-project.org/package=lm.beta

Belaire, J. A., L. M. Westphal, and E. S. Minor. 2016. Different social drivers, including perceptions of urban wildlife, explain the ecological resources in residential landscapes. Landscape Ecology 31:401–413.

Bertocci, I., F. Arenas, E. Cacabelos, G. M. Martins, M. I. Seabra, N. V. Álvaro, J. N. Fernandes, R. Gaião, N. Mamede, M. Mulas, and A. I. Neto. 2017. Nowhere safe? Exploring the influence of urbanization across mainland and insular seashores in continental Portugal and the Azorean Archipelago. Marine Pollution Bulletin 114:644–655.

Bettez, N. D., and P. M. Groffman. 2013. Nitrogen deposition in and near an urban ecosystem. Environmental Science & Technology 47:6047–6051.

Bharath, S., E. T. Borer, L. A. Biederman, D. M. Blumenthal, P. A. Fay, L. A. Gherardi, J. M. H. Knops, A. D. B. Leakey, L. Yahdjian, and E. W. Seabloom. 2020. Nutrient addition increases grassland sensitivity to droughts. Ecology 101:e02981.

Bigsby, K. M., M. R. McHale, and G. R. Hess. 2014. Urban morphology drives the homogenization of tree cover in Baltimore, MD, and Raleigh, NC. Ecosystems 17:212–227.

Blaine, T. W., S. Clayton, P. Robbins, and P. S. Grewal. 2012. Homeowner attitudes and practices towards residential landscape management in Ohio, USA. Environmental Management 50:257–271.

Blair, R. 2004. The effects of urban sprawl on birds at multiple levels of biological organization. Ecology and Society 9:2.

Blair, R. B., and E. M. Johnson. 2008. Suburban habitats and their role for birds in the urban-rural habitat network: points of local invasion and extinction? Landscape Ecology 23:1157–1169.

Bobbink, R., K. Hicks, J. Galloway, T. Spranger, R. Alkemade, M. Ashmore, M. Bustamante, S. Cinderby, E. Davidson, F. Dentener, B. Emmett, J.-W. Erisman, M. Fenn, F. Gilliam, A. Nordin, L. Pardo, and W. De Vries. 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. Ecological Applications 20:30–59.

Bohrer, V. L. 1991. Recently recognized cultivated and encouraged plants among the Hohokam. Kiva 56:227–235.

Borer, E. T., E. W. Seabloom, D. S. Gruner, W. S. Harpole, H. Hillebrand, E. M. Lind, P. B. Adler, J. Alberti, T. M. Anderson, J. D. Bakker, L. Biederman, D. Blumenthal, C. S. Brown, L. A. Brudvig, Y. M. Buckley, M. Cadotte, C. Chu, E. E. Cleland, M. J. Crawley, P. Daleo, E. I. Damschen, K. F. Davies, N. M. DeCrappeo, G. Du, J. Firn, Y. Hautier, R. W. Heckman, A. Hector, J. HilleRisLambers, O. Iribarne, J. A. Klein, J. M. H. Knops, K. J. La Pierre, A. D. B. Leakey, W. Li, A. S. MacDougall, R. L. McCulley, B. A. Melbourne, C. E. Mitchell, J. L. Moore, B. Mortensen, L. R. O’Halloran, J. L. Orrock, J. Pascual, S. M. Prober, D. A. Pyke, A. C. Risch, M.

Schuetz, M. D. Smith, C. J. Stevens, L. L. Sullivan, R. J. Williams, P. D. Wragg, J. P. Wright, and L. H. Yang. 2014. Herbivores and nutrients control grassland plant diversity via light limitation. Nature 508:517–520.

Bowers, J. E. 2005. El Niño and displays of spring-flowering annuals in the Mojave and Sonoran deserts. The Journal of the Torrey Botanical Society 132:38–49.

Bowers, J. E., T. M. Bean, and R. M. Turner. 2006. Two decades of change in distribution of exotic plants at the Desert Laboratory, Tucson, Arizona. Madroño 53:252–263.

Bradley, B. A., D. M. Blumenthal, R. Early, E. D. Grosholz, J. J. Lawler, L. P. Miller, C. J. B. Sorte, C. M. D'Antonio, J. M. Diez, J. S. Dukes, I. Ibanez, and J. D. Olden. 2012. Global change, global trade, and the next wave of plant invasions. Frontiers in Ecology and the Environment 10:20–28.

Brice, M.-H., S. Pellerin, and M. Poulin. 2017. Does urbanization lead to taxonomic and functional homogenization in riparian forests? Diversity and Distributions 23:828–840.

Brooks, M. L. 2003. Effects of increased soil nitrogen on the dominance of alien annual plants in the Mojave Desert. Journal of Applied Ecology 40:344–353.

Brzuszek, R. F., and R. L. Harkess. 2009. Green industry survey of native plant marketing in the southeastern United States. HortTechnology 19:168–172.

Brzuszek, R. F., R. L. Harkess, and L. Kelly. 2010. Survey of master gardener use of native plants in the southeastern United States. HortTechnology 20:462–466.

Brzuszek, R. F., R. L. Harkess, and S. J. Mulley. 2007. Landscape architects' use of native plants in the southeastern United States. HortTechnology 17:78–81.

Burghardt, K. T., D. W. Tallamy, and W. G. Shriver. 2009. Impact of native plants on bird and butterfly biodiversity in suburban landscapes. Conservation Biology 23:219–224.

Burnham, K. P., and D. R. Anderson. 2004. Multimodel inference: understanding AIC and BIC in model selection. Sociological Methods & Research 33:261–304.

Burr, A., D. M. Hall, and N. Schaeg. 2018. The perfect lawn: exploring neighborhood socio-cultural drivers for insect pollinator habitat. Urban Ecosystems 21:1123–1137.

Butchart, S. H. M., M. Walpole, B. Collen, A. van Strien, J. P. W. Scharlemann, R. E. A. Almond, J. E. M. Baillie, B. Bomhard, C. Brown, J. Bruno, K. E. Carpenter, G. M. Carr, J. Chanson, A. M. Chenery, J. Csirke, N. C. Davidson, F. Dentener, M. Foster,

A. Galli, J. N. Galloway, P. Genovesi, R. D. Gregory, M. Hockings, V. Kapos, J.-F. Lamarque, F. Leverington, J. Loh, M. A. McGeoch, L. McRae, A. Minasyan, M. Hernández Morcillo, T. E. E. Oldfield, D. Pauly, S. Quader, C. Revenga, J. R. Sauer, B. Skolnik, D. Spear, D. Stanwell-Smith, S. N. Stuart, A. Symes, M. Tierney, T. D. Tyrrell, J.-C. Vié, and R. Watson. 2010. Global biodiversity: indicators of recent declines. Science 328:1164–1168.

Butterfield, B. J., and R. M. Callaway. 2013. A functional comparative approach to facilitation and its context dependence. Functional Ecology 27:907–917.

Cadotte, M. W., S. L. E. Yasui, S. Livingstone, and J. S. MacIvor. 2017. Are urban systems beneficial, detrimental, or indifferent for biological invasion? Biological Invasions 19:3489–3503.

Cardinale, B. J., J. E. Duffy, A. Gonzalez, D. U. Hooper, C. Perrings, P. Venail, A. Narwani, G. M. Mace, D. Tilman, D. A. Wardle, A. P. Kinzig, G. C. Daily, M. Loreau, J. B. Grace, A. Larigauderie, D. S. Srivastava, and S. Naeem. 2012. Biodiversity loss and its impact on humanity. Nature 486:59–67.

Catterall, C. P., J. A. Cousin, S. Piper, and G. Johnson. 2010. Long-term dynamics of bird diversity in forest and suburb: decay, turnover or homogenization? Diversity and Distributions 16:559–570.

Cavender-Bares, J., K. H. Kozak, P. V. A. Fine, and S. W. Kembel. 2009. The merging of community ecology and phylogenetic biology. Ecology Letters 12:693–715.

Cavender-Bares, J., J. Padullés Cubino, W. D. Pearse, S. E. Hobbie, A. J. Lange, S. Knapp, and K. C. Nelson. 2020. Horticultural availability and homeowner preferences drive plant diversity and composition in urban yards. Ecological Applications 30:e02082.

Chapin, F. S. III, E. S. Zavaleta, V. T. Eviner, R. L. Naylor, P. M. Vitousek, H. L. Reynolds, D. U. Hooper, S. Lavorel, O. E. Sala, S. E. Hobbie, M. C. Mack, and S. Díaz. 2000. Consequences of changing biodiversity. Nature 405:234–242.

Childers, D. L., N. B. Grimm, D. Hope, J. P. Kaye, C. A. Martin, N. E. McIntyre, and J. Stutz. 2018. Ecological Survey of Central Arizona: a survey of key ecological indicators in the greater Phoenix metropolitan area and surrounding Sonoran Desert, ongoing since 1999. https://sustainability.asu.edu/caplter/data/view/knb-lter-cap.652/.

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

Clements, F. E. 1916. Plant succession: an analysis of the development of vegetation. Carnegie Institution of Washington.

Collins, S. L., S. R. Carpenter, S. M. Swinton, D. E. Orenstein, D. L. Childers, T. L. Gragson, N. B. Grimm, J. M. Grove, S. L. Harlan, J. P. Kaye, A. K. Knapp, G. P. Kofinas, J. J. Magnuson, W. H. McDowell, J. M. Melack, L. A. Ogden, G. P. Robertson, M. D. Smith, and A. C. Whitmer. 2011. An integrated conceptual framework for long-term social-ecological research. Frontiers in Ecology and the Environment 9:351–357.

Conway, T. M. 2016. Tending their urban forest: residents' motivations for tree planting and removal. Urban Forestry & Urban Greening 17:23–32.

Conway, T. M., and K. S. Bourne. 2013. A comparison of neighborhood characteristics related to canopy cover, stem density and species richness in an urban forest. Landscape and Urban Planning 113:10–18.

Conway, T. M., and J. Vander Vecht. 2015. Growing a diverse urban forest: species selection decisions by practitioners planting and supplying trees. Landscape and Urban Planning 138:1–10.

Conway, T. M., and V. Yip. 2016. Assessing residents' reactions to urban forest disservices: a case study of a major storm event. Landscape and Urban Planning 153:1–10.

Cook, E. M., S. J. Hall, and K. L. Larson. 2012. Residential landscapes as social-ecological systems: a synthesis of multi-scalar interactions between people and their home environment. Urban Ecosystems 15:19–52.

Cook, E. M., R. Sponseller, N. B. Grimm, and S. J. Hall. 2018. Mixed method approach to assess atmospheric nitrogen deposition in arid and semi-arid ecosystems. Environmental Pollution 239:617–630.

Cox, D. T. C., H. L. Hudson, D. F. Shanahan, R. A. Fuller, and K. J. Gaston. 2017. The rarity of direct experiences of nature in an urban population. Landscape and Urban Planning 160:79–84.

Cox, D. T. C., D. F. Shanahan, H. L. Hudson, R. A. Fuller, and K. J. Gaston. 2018. The impact of urbanisation on nature dose and the implications for human health. Landscape and Urban Planning 179:72–80.

Crewe, K. 2013. Arizona native plants and the urban challenge. Landscape Journal 32:215–229.

Davis, M. K., E. M. Cook, S. L. Collins, and S. J. Hall. 2015. Top-down vs. bottom-up regulation of herbaceous primary production and composition in an arid, urbanizing ecosystem. Journal of Arid Environments 116:103–114.

Deák, B., B. Hüse, and B. Tóthmérész. 2016. Grassland vegetation in urban habitats - testing ecological theories. Tuexenia 36:379–393.

Dearborn, D. C., and S. Kark. 2010. Motivations for conserving urban biodiversity. Conservation Biology 24:432–440.

Derby Lewis, A., M. J. Bouman, A. M. Winter, E. A. Hasle, D. F. Stotz, M. K. Johnston, K. R. Klinger, A. Rosenthal, and C. A. Czarnecki. 2019. Does nature need cities? Pollinators reveal a role for cities in wildlife conservation. Frontiers in Ecology and Evolution 7:220.

Dolan, R. W., M. E. Moore, and J. D. Stephens. 2011. Documenting effects of urbanization on flora using herbarium records. Journal of Ecology 99:1055–1062.

Dornelas, M., N. J. Gotelli, H. Shimadzu, F. Moyes, A. E. Magurran, and B. J. McGill. 2019. A balance of winners and losers in the Anthropocene. Ecology Letters 22:847–854.

Drescher, M., G. K. Warriner, J. R. Farmer, and B. M. H. Larson. 2017. Private landowners and environmental conservation: a case study of social- psychological determinants of conservation program participation in Ontario. Ecology and Society 22:44.

Duncan, R. P., S. E. Clemants, R. T. Corlett, A. K. Hahs, M. A. McCarthy, M. J. McDonnell, M. W. Schwartz, K. Thompson, P. A. Vesk, and N. S. G. Williams. 2011. Plant traits and extinction in urban areas: a meta-analysis of 11 cities. Global Ecology and Biogeography 20:509–519.

Ekroos, J., J. Heliölä, and M. Kuussaari. 2010. Homogenization of lepidopteran communities in intensively cultivated agricultural landscapes. Journal of Applied Ecology 47:459–467.

Elser, J. J., M. E. S. Bracken, E. E. Cleland, D. S. Gruner, W. S. Harpole, H. Hillebrand, J. T. Ngai, E. W. Seabloom, J. B. Shurin, and J. E. Smith. 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecology Letters 10:1135–1142.

Epp Schmidt, D. J., R. Pouyat, K. Szlavecz, H. Setälä, D. J. Kotze, I. Yesilonis, S. Cilliers, E. Hornung, M. Dombos, and S. A. Yarwood. 2017. Urbanization erodes ectomycorrhizal fungal diversity and may cause microbial communities to converge. Nature Ecology & Evolution 1:0123.

Epskamp, S. 2019. semPlot: path diagrams and visual analysis of various SEM packages' output. R package version 1.1.2. https://CRAN.R-project.org/package=semPlot

Erisman, J. W., J. N. Galloway, S. Seitzinger, A. Bleeker, N. B. Dise, A. M. R. Petrescu, A. M. Leach, and W. de Vries. 2013. Consequences of human modification of the global nitrogen cycle. Philosophical Transactions of the Royal Society B 368:20130116.

Facelli, J. M., and A. M. Temby. 2002. Multiple effects of shrubs on annual plant communities in arid lands of South Australia. Austral Ecology 27:422–432.

Fay, P. A., S. M. Prober, W. S. Harpole, J. M. H. Knops, J. D. Bakker, E. T. Borer, E. M. Lind, A. S. MacDougall, E. W. Seabloom, P. D. Wragg, P. B. Adler, D. M. Blumenthal, Y. M. Buckley, C. Chu, E. E. Cleland, S. L. Collins, K. F. Davies, G. Du, X. Feng, J. Firn, D. S. Gruner, N. Hagenah, Y. Hautier, R. W. Heckman, V. L. Jin, K. P. Kirkman, J. Klein, L. M. Ladwig, Q. Li, R. L. McCulley, B. A. Melbourne, C. E. Mitchell, J. L. Moore, J. W. Morgan, A. C. Risch, M. Schütz, C. J. Stevens, D. A. Wedin, and L. H. Yang. 2015. Grassland productivity limited by multiple nutrients. Nature Plants 1:15080.

Fenn, M. E., J. S. Baron, E. B. Allen, H. M. Rueth, K. R. Nydick, L. Geiser, W. D. Bowman, J. O. Sickman, T. Meixner, D. W. Johnson, and P. Neitlich. 2003a. Ecological effects of nitrogen deposition in the western United States. BioScience 53:404–420.

Fenn, M. E., R. Haeuber, G. S. Tonnesen, J. S. Baron, S. Grossman-Clarke, D. Hope, D. A. Jaffe, S. Copeland, L. Geiser, H. M. Rueth, and J. O. Sickman. 2003b. Nitrogen emissions, deposition, and monitoring in the western United States. BioScience 53:391–403.

Field, C. D., N. B. Dise, R. J. Payne, A. J. Britton, B. A. Emmett, R. C. Helliwell, S. Hughes, L. Jones, S. Lees, J. R. Leake, I. D. Leith, G. K. Phoenix, S. A. Power, L. J. Sheppard, G. E. Southon, C. J. Stevens, and S. J. M. Caporn. 2014. The role of nitrogen deposition in widespread plant community change across semi-natural habitats. Ecosystems 17:864–877.

Filloy, J., S. Grosso, and M. I. Bellocq. 2015. Urbanization altered latitudinal patterns of bird diversity-environment relationships in the southern Neotropics. Urban Ecosystems 18:777–791.

Fischer, A., B. Bednar-Friedl, F. Langers, M. Dobrovodská, N. Geamana, K. Skogen, and M. Dumortier. 2011. Universal criteria for species conservation priorities? Findings from a survey of public views across Europe. Biological Conservation 144:998–1007.

Fischer, L. K., V. Rodorff, M. von der Lippe, and I. Kowarik. 2016. Drivers of biodiversity patterns in parks of a growing South American megacity. Urban Ecosystems 19:1231–1249.

Flood Control District of Maricopa County. 2018. Weather sensor data. https://www.maricopa.gov/3769/Weather-Sensor-Data.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. III Chapin, M. T. Coe, G. C. Daily, H. K. Gibbs, J. H. Helkowski, T. Holloway, E. A. Howard, C. J. Kucharik, C. Monfreda, J. A. Patz, I. C. Prentice, N. Ramankutty, and P. K. Snyder. 2005. Global consequences of land use. Science 309:570–574.

Frick, J., F. G. Kaiser, and M. Wilson. 2004. Environmental knowledge and conservation behavior: exploring prevalence and structure in a representative sample. Personality and Individual Differences 37:1597–1613.

Frost, D. 2016, January. An Eye on Every Drop. American Planning Association Magazine.

Fuller, R. A., K. N. Irvine, P. Devine-Wright, P. H. Warren, and K. J. Gaston. 2007. Psychological benefits of greenspace increase with biodiversity. Biology Letters 3:390–394.

Galloway, J. N., A. R. Townsend, J. W. Erisman, M. Bekunda, Z. Cai, J. R. Freney, L. A. Martinelli, S. P. Seitzinger, and M. A. Sutton. 2008. Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. Science 320:889–892.

Gamo, M., M. Shinoda, and T. Maeda. 2013. Classification of arid lands, including soil degradation and irrigated areas, based on vegetation and aridity indices. International Journal of Remote Sensing 34:6701–6722.

Garcillán, P. P., E. D. Dana, J. P. Rebman, and J. Peñas. 2014. Effects of alien species on homogenization of urban floras across continents: a tale of two mediterranean cities in two different continents. Plant Ecology and Evolution 147:3–9.

Gaston, K. J., M. L. Ávila-Jiménez, and J. L. Edmondson. 2013. Managing urban ecosystems for goods and services. Journal of Applied Ecology 50:830–840.

Ge, X.-Y. M., J. P. Scholl, U. Basinger, T. E. Huxman, and D. L. Venable. 2019. Functional trait trade-off and species abundance: insights from a multi-decadal study. Ecology Letters 22:583–592.

Gerrish, E., and S. L. Watkins. 2018. The relationship between urban forests and income: a meta-analysis. Landscape and Urban Planning 170:293–308.

Goddard, M. A., A. J. Dougill, and T. G. Benton. 2009. Scaling up from gardens: biodiversity conservation in urban environments. Trends in Ecology & Evolution 25:90–98.

Goddard, M. A., A. J. Dougill, and T. G. Benton. 2013. Why garden for wildlife? Social and ecological drivers, motivations and barriers for biodiversity management in residential landscapes. Ecological Economics 86:258–273.

Gregg, J. W., C. G. Jones, and T. E. Dawson. 2003. Urbanization effects on tree growth in the vicinity of New York City. Letters to Nature 424:183–187.

Gremer, J. R., S. Kimball, A. L. Angert, D. L. Venable, and T. E. Huxman. 2012. Variation in photosynthetic response to temperature in a guild of winter annual plants. Ecology 93:2693–2704.

Gremer, J. R., S. Kimball, K. R. Keck, T. E. Huxman, A. L. Angert, and D. L. Venable. 2013. Water-use efficiency and relative growth rate mediate competitive interactions in Sonoran Desert winter annual plants. American Journal of Botany 100:2009–2015.

Gremer, J. R., S. Kimball, and D. L. Venable. 2016. Within- and among-year germination in Sonoran Desert winter annuals: bet hedging and predictive germination in a variable environment. Ecology Letters 19:1209–1218.

Gremer, J. R., and D. L. Venable. 2014. Bet hedging in desert winter annual plants: optimal germination strategies in a variable environment. Ecology Letters 17:380–387.

Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai, and J. M. Briggs. 2008. Global change and the ecology of cities. Science 319:756–760.

Grimm, N. B., S. J. Hall, J. P. Kaye, J. Allen, and D. L. Childers. 2017. CAP weather stations at Papago Park and Lost Dutchman State Park in the greater Phoenix metropolitan area, ongoing since 2010. https://sustainability.asu.edu/caplter/data/data-catalog/view/knb-lter-cap.636/.

Grimm, N. B., S. J. Hall, J. P. Kaye, J. Allen, and D. L. Childers. 2019. Desert fertilization experiment: investigation of Sonoran Desert ecosystem response to atmospheric deposition and experimental nutrient addition, ongoing since 2006. https://sustainability.asu.edu/caplter/data/data-catalog/view/knb-lter-cap.632/.

Groffman, P. M., M. Avolio, J. Cavender-Bares, N. D. Bettez, J. M. Grove, S. J. Hall, S. E. Hobbie, K. L. Larson, S. B. Lerman, D. H. Locke, J. B. Heffernan, J. L. Morse, C. Neill, K. C. Nelson, J. O'Neil-Dunne, D. E. Pataki, C. Polsky, R. Roy Chowdhury, and T. L. E. Trammell. 2017. Ecological homogenization of residential macrosystems. Nature Ecology & Evolution 1:0191.

Groffman, P. M., J. Cavender-Bares, N. D. Bettez, J. M. Grove, S. J. Hall, J. B. Heffernan, S. E. Hobbie, K. L. Larson, J. L. Morse, C. Neill, K. Nelson, J. O'Neil-Dunne, L. Ogden, D. E. Pataki, C. Polsky, R. Roy Chowdhury, and M. K. Steele. 2014. Ecological homogenization of urban USA. Frontiers in Ecology and the Environment 12:74–81.

Groffman, P. M., J. M. Grove, C. Polsky, N. D. Bettez, J. L. Morse, J. Cavender-Bares, S. J. Hall, J. B. Heffernan, S. E. Hobbie, K. L. Larson, C. Neill, K. Nelson, L. Ogden, J. O'Neil-Dunne, D. Pataki, R. Roy Chowdhury, and D. H. Locke. 2016. Satisfaction, water and fertilizer use in the American residential macrosystem. Environmental Research Letters 11:034004.

Grove, J. M., D. H. Locke, and J. P. M. O'Neil-Dunne. 2014. An ecology of prestige in New York City: examining the relationships among population density, socio-economic status, group identity, and residential canopy cover. Environmental Management 54:402–419.

Grove, M., L. Ogden, S. Pickett, C. Boone, G. Buckley, D. H. Locke, C. Lord, and B. Hall. 2017. The legacy effect: understanding how segregation and environmental injustice unfold over time in Baltimore. Annals for the American Association of Geographers 108:524–537.

Guo, T., J. Morgenroth, and T. Conway. 2018. Redeveloping the urban forest: the effect of redevelopment and property-scale variables on tree removal and retention. Urban Forestry & Urban Greening 35:192–201.

Hadley, N. F., and S. R. Szarek. 1981. Productivity of desert ecosystems. BioScience 31:747–753.

Hall, S. J., J. Learned, B. Ruddell, K. L. Larson, J. Cavender-Bares, N. Bettez, P. M. Groffman, J. M. Grove, J. B. Heffernan, S. E. Hobbie, J. L. Morse, C. Neill, K. C. Nelson, J. P. M. O'Neil-Dunne, L. Ogden, D. E. Pataki, W. D. Pearse, C. Polsky, R. Roy Chowdhury, M. K. Steele, and T. L. E. Trammell. 2016. Convergence of microclimate in residential landscapes across diverse cities in the United States. Landscape Ecology 31:101–117.

Hall, S. J., R. A. Sponseller, N. B. Grimm, D. Huber, J. P. Kaye, C. Clark, and S. L. Collins. 2011. Ecosystem response to nutrient enrichment across an urban airshed in the Sonoran Desert. Ecological Applications 21:640–660.

Hargreaves, G. H., and R. G. Allen. 2003. History and evaluation of Hargreaves evapotranspiration equation. Journal of Irrigation and Drainage Engineering 129:53–63.

Harpole, W. S., L. L. Sullivan, E. M. Lind, J. Firn, P. B. Adler, E. T. Borer, J. Chase, P. A. Fay, Y. Hautier, H. Hillebrand, A. S. MacDougall, E. W. Seabloom, R. Williams, J. D. Bakker, M. W. Cadotte, E. J. Chaneton, C. Chu, E. E. Cleland, C. D'Antonio, K. F. Davies, D. S. Gruner, N. Hagenah, K. Kirkman, J. M. H. Knops, K. J. La Pierre, R. L. McCulley, J. L. Moore, J. W. Morgan, S. M. Prober, A. C. Risch, M. Schuetz, C. J. Stevens, and P. D. Wragg. 2016. Addition of multiple limiting resources reduces grassland diversity. Nature 537:93–96.

Harris, E. M., C. Polsky, K. L. Larson, R. Garvoille, D. G. Martin, J. Brumand, and L. Ogden. 2012. Heterogeneity in residential yard care: evidence from Boston, Miami, and Phoenix. Human Ecology 40:735–749.

Harrison, T., J. Gibbs, and R. Winfree. 2018. Phylogenetic homogenization of bee communities across ecoregions. Global Ecology and Biogeography 27:1457–1466.

Hartig, T., R. Mitchell, S. de Vries, and H. Frumkin. 2014. Nature and Health. Annual Review of Public Health 35:207–228.

Hartley, D. J., M. J. Koivula, J. R. Spence, R. Pelletier, and G. E. Ball. 2007. Effects of urbanization on ground beetle assemblages (Coleoptera, Carabidae) of grassland habitats in western Canada. Ecography 30:673–684.

Hausmann, A., R. Slotow, J. K. Burns, and E. Di Minin. 2016. The ecosystem service of sense of place: benefits for human well-being and biodiversity conservation. Environmental Conservation 43:117–127.

Hautier, Y., P. A. Niklaus, and A. Hector. 2009. Competition for light causes plant biodiversity loss after eutrophication. Science 324:636–638.

Head, L., and P. Muir. 2004. Nativeness, invasiveness, and nation in Australian plants. The Geographical Review 94:199–217.

Head, L., and P. Muir. 2006. Suburban life and the boundaries of nature: resilience and rupture in Australian backyard gardens. Transactions of the Institute of British Geographers 31:505–524.

Heavenrich, H., and S. J. Hall. 2016. Elevated soil nitrogen pools after conversion of turfgrass to water-efficient residential landscapes. Environmental Research Letters 11:084007.

Heberlein, T. A. 2012. Navigating environmental attitudes. Oxford University Press, New York, New York, USA.

Heberlein, T. A. 2013. Navigating environmental attitudes. Conservation Biology 26:583–585.

Hirt, P., A. Gustafson, and K. L. Larson. 2008. The mirage in the valley of the sun. Environmental History 13:482–514.

Holway, D. A., and A. V. Suarez. 2006. Homogenization of ant communities in mediterranean California: the effects of urbanization and invasion. Biological Conservation 127:319–326.

Holzapfel, C., and B. E. Mahall. 1999. Bidirectional facilitation and interference between shrubs and annuals in the Mojave Desert. Ecology 80:1747–1761.

Hooper, D. U., and L. Johnson. 1999. Nitrogen limitation in dryland ecosystems: responses to geographical and temporal variation in precipitation. Biogeochemistry 46:247–293.

Hooper, V. H., J. Endter-Wada, and C. W. Johnson. 2008. Theory and practice related to native plants: a case study of Utah landscape professionals. Landscape Journal 27:127–141.

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

Horsák, M., T. Čejka, L. Juřičková, V. Wiese, V. Horsáková, and Z. Lososová. 2016. Drivers of Central European urban land snail faunas: the role of climate and local species pool in the representation of native and non-native species. Biological Invasions 18:3547–3560.

Horsák, M., Z. Lososová, T. Čejka, L. Juřičková, and M. Chytrý. 2013. Diversity and biotic homogenization of urban land-snail faunas in relation to habitat types and macroclimate in 32 Central European cities. PLoS ONE 8:e71783.

Hoyle, H., J. Hitchmough, and A. Jorgensen. 2017. Attractive, climate-adapted and sustainable? Public perception of non-native planting in the designed urban landscape. Landscape and Urban Planning 164:49–63.

Huang, Q., D. T. Robinson, and D. C. Parker. 2014. Quantifying spatial-temporal change in land-cover and carbon storage among exurban residential parcels. Landscape Ecology 29:275–291.

Hunter, M. C. R., and D. G. Brown. 2012. Spatial contagion: gardening along the street in residential neighborhoods. Landscape and Urban Planning 105:407–416.

Huwer, A., and R. Wittig. 2013. Evidence for increasing homogenization and de-ruralization of the Central European village flora. Tuexenia 33:213–231.

Huxman, T. E., G. Barron-Gafford, K. L. Gerst, A. L. Angert, A. P. Tyler, and D. L. Venable. 2008. Photosynthetic resource-use efficiency and demographic variability in desert winter annual plants. Ecology 89:1554–1563.

Huxman, T. E., S. Kimball, A. L. Angert, J. R. Gremer, G. A. Barron-Gafford, and D. L. Venable. 2013. Understanding past, contemporary, and future dynamics of plants, populations, and communities using Sonoran Desert winter annuals. American Journal of Botany 100:1369–1380.

Imhoff, M. L., P. Zhang, R. E. Wolfe, and L. Bounoua. 2010. Remote sensing of the urban heat island effect across biomes in the continental USA. Remote Sensing of Environment 114:504–513.

Ives, C. D., P. E. Lentini, C. G. Threlfall, K. Ikin, D. F. Shanahan, G. E. Garrard, S. A. Bekessy, R. A. Fuller, L. Mumaw, L. Rayner, R. Rowe, L. E. Valentine, and D. Kendal. 2016. Cities are hotspots for threatened species. Global Ecology and Biogeography 25:117–126.

Iwaniec, D. M., E. M. Cook, M. J. Davidson, M. Berbés-Blázquez, and N. B. Grimm. 2020. Integrating existing climate adaptation planning into future visions: a strategic scenario for the central Arizona–Phoenix region. Landscape and Urban Planning 200:103820.

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

Jenerette, G. D., J. Wu, N. B. Grimm, and D. Hope. 2006. Points, patches, and regions: scaling soil biogeochemical patterns in an urbanized arid ecosystem. Global Change Biology 12:1532–1544.

Jennings, W. B., and K. H. Berry. 2015. Desert tortoises (*Gopherus agassizii*) are selective herbivores that track the flowering phenology of their preferred food plants. PLoS ONE 10:e0116716.

Jim, C. Y., and W. Y. Chen. 2008. Pattern and divergence of tree communities in Taipei's main urban green spaces. Landscape and Urban Planning 84:312–323.

Johnson, A. L., D. Borowy, and C. M. Swan. 2018. Land use history and seed dispersal drive divergent plant community assembly patterns in urban vacant lots. Journal of Applied Ecology 55:451–460.

Johnson, P. T. J., J. T. Hoverman, V. J. McKenzie, A. R. Blaustein, and K. L. D. Richgels. 2013. Urbanization and wetland communities: applying metacommunity theory to understand the local and landscape effects. Journal of Applied Ecology 50:34–42.

Kale, M., N. Dudhe, M. Ferrante, T. Ivanova, R. Kasambe, I. S. Trukhanova, P. Bhattacharya, and G. L. Lövei. 2018. The effect of urbanization on the functional and scale-sensitive diversity of bird assemblages in Central India. Journal of Tropical Ecology 34:341–350.

Kartesz, J. T. 2015. North American plant atlas. The Biota of North America Program (BONAP), Chapel Hill, North Carolina, USA. http://bonap.net/napa.

Kauth, P. J., and H. E. Pérez. 2011. Industry survey of the native wildflower market in Florida. HortTechnology 21:779–788.

Kaye, J. P., P. M. Groffman, N. B. Grimm, L. A. Baker, and R. V. Pouyat. 2006. A distinct urban biogeochemistry? Trends in Ecology & Evolution 21:192–199.

Kembel, S. W., P. D. Cowan, M. R. Helmus, W. K. Cornwell, H. Morlon, D. D. Ackerly, S. P. Blomberg, and C. O. Webb. 2010. Picante: R tools for integrating phylogenies and ecology. Bioinformatics 26:1463–1464.

Kendal, D., K. J. H. Williams, and N. S. G. Williams. 2012. Plant traits link people's plant preferences to the composition of their gardens. Landscape and Urban Planning 105:34–42.

Kendle, A. D., and J. E. Rose. 2000. The aliens have landed! What are the justifications for "native only" policies in landscape plantings? Landscape and Urban Planning 47:19–31.

Kenny, D. A., B. Kaniskan, and D. B. McCoach. 2015. The performance of RMSEA in models with small degrees of freedom. Sociological Methods & Research 44:486–507.

Kim, M., and J. E. Hunter. 1993. Attitude-behavior relations: a meta-analysis of attitudinal relevance and topic. Journal of Communication 43:101–142.

Kimball, S., A. L. Angert, T. E. Huxman, and D. L. Venable. 2010. Contemporary climate change in the Sonoran Desert favors cold-adapted species. Global Change Biology 16:1555–1565.

Kimball, S., A. L. Angert, T. E. Huxman, and D. L. Venable. 2011. Differences in the timing of germination and reproduction relate to growth physiology and population dynamics of Sonoran Desert winter annuals. American Journal of Botany 98:1773–1781.

Kimball, S., J. R. Gremer, A. L. Angert, T. E. Huxman, and D. L. Venable. 2012. Fitness and physiology in a variable environment. Oecologia 169:319–329.

Kimball, S., J. R. Gremer, G. A. Barron-Gafford, A. L. Angert, T. E. Huxman, and D. L. Venable. 2014. High water-use efficiency and growth contribute to success of non-native *Erodium cicutarium* in a Sonoran Desert winter annual community. Conservation Physiology 2.

Kirkpatrick, J. B., A. Davison, and G. D. Daniels. 2013. Sinners, scapegoats or fashion victims? Understanding the deaths of trees in the green city. Geoforum 48:165–176.

Knapp, S., I. Kühn, R. Wittig, W. A. Ozinga, P. Poschlod, and S. Klotz. 2008. Urbanization causes shifts in species' trait state frequencies. Preslia 80:375–388.

Knapp, S., and R. Wittig. 2012. An analysis of temporal homogenisation and differentiation in Central European village floras. Basic and Applied Ecology 13:319–327.

Knop, E. 2016. Biotic homogenization of three insect groups due to urbanization. Global Change Biology 22:228–236.

Ko, Y., J.-H. Lee, E. G. McPherson, and L. A. Roman. 2015. Factors affecting long-term mortality of residential shade trees: evidence from Sacramento, California. Urban Forestry & Urban Greening 14:500–507.

Kollmuss, A., and J. Agyeman. 2002. Mind the gap: why do people behave environmentally and what are the barriers to pro-environmental behavior? Environmental Education Research 8:239–260.

Koricheva, J., J. Gurevitch, and K. Mengersen, editors. 2013. Handbook of meta-analysis in ecology and evolution. Princeton University Press.

Korner-Nievergelt, F., T. Roth, S. von Felten, J. Guelat, B. Almasi, and P. Korner-Nievergelt. 2015. blmeco: data files and functions accompanying the book "Bayesian data analysis in ecology using linear models with R, BUGS and Stan." R package version 1.4. https://CRAN.R-project.org/package=blmeco

La Sorte, F. A., M. F. J. Aronson, N. S. G. Williams, L. Celesti-Grapow, S. Cilliers, B. D. Clarkson, R. W. Dolan, A. Hipp, S. Klotz, I. Kühn, P. Pyšek, S. Siebert, and M. Winter. 2014a. Beta diversity of urban floras among European and non-European cities. Global Ecology and Biogeography 23:769–779.

La Sorte, F. A., and M. L. McKinney. 2006. Compositional similarity and the distribution of geographical range size for assemblages of native and non-native species in urban floras. Diversity and Distributions 12:679–686.

La Sorte, F. A., M. L. McKinney, and P. Pyšek. 2007. Compositional similarity among urban floras within and across continents: biogeographical consequences of human-mediated biotic interchange. Global Change Biology 13:913–921.

La Sorte, F. A., M. L. McKinney, P. Pyšek, S. Klotz, G. L. Rapson, L. Celesti-Grapow, and K. Thompson. 2008. Distance decay of similarity among European urban floras: the impact of anthropogenic activities on β diversity. Global Ecology and Biogeography 17:363–371.

La Sorte, F. A., M. W. Tingley, and A. H. Hurlbert. 2014b. The role of urban and agricultural areas during avian migration: an assessment of within-year temporal turnover. Global Ecology and Biogeography 23:1225–1234.

Ladwig, L. M., S. L. Collins, A. L. Swann, Y. Xia, M. F. Allen, and E. B. Allen. 2012. Above- and belowground responses to nitrogen addition in a Chihuahuan Desert grassland. Oecologia 169:177–185.

Laiolo, P. 2011. Homogenisation of birdsong along a natural-urban gradient in Argentina. Ethology Ecology & Evolution 23:274–287.

Landry, S. M., and J. Chakraborty. 2009. Street trees and equity: evaluating the spatial distribution of an urban amenity. Environment and Planning A 41:2651–2670.

Landscape plants for the Arizona desert. 2006. Arizona Municipal Water Users Association (AMWUA). http://www.amwua.org/plants.

Larsen, L., and S. L. Harlan. 2006. Desert dreamscapes: residential landscape preference and behavior. Landscape and Urban Planning 78:85–100.

Larson, K. L. 2010. An integrated theoretical approach to understanding the sociocultural basis of multidimensional environmental attitudes. Society & Natural Resources 23:898–907.

Larson, K. L., R. Andrade, A. York, D. Childers, P. Coseo, S. Earl, M. Ehlenz, N. Grimm, S. J. Hall, S. Harlan, D. Hondula, S. B. Lerman, D. Pfeiffer, J. Ripplinger, K. Smith, P. Warren, M. Watkins, D. White, S. Wittlinger, M. Wright, and A. Wutich. 2017a. The Phoenix Area Social Survey IV: Linking Social and Biophysical Dynamics in Urban Neighborhoods. A report from the Central Arizona–Phoenix Long-Term Ecological Research (CAP LTER) project. Tempe, Arizona.

Larson, K. L., and J. Brumand. 2014. Paradoxes in landscape management and water conservation: examining neighborhood norms and institutional forces. Cities and the Environment 7:6.

Larson, K. L., D. Casagrande, S. L. Harlan, and S. T. Yabiku. 2009. Residents' yard choices and rationales in a desert city: social priorities, ecological impacts, and decision tradeoffs. Environmental Management 44:921–937.

Larson, K. L., E. Cook, C. Strawhacker, and S. J. Hall. 2010. The influence of diverse values, ecological structure, and geographic context on residents' multifaceted landscaping decisions. Human Ecology 38:747–761.

Larson, K. L., J. Hoffman, and J. Ripplinger. 2017b. Legacy effects and landscape choices in a desert city. Landscape and Urban Planning 165:22–29.

Larson, K. L., K. C. Nelson, S. R. Samples, S. J. Hall, N. Bettez, J. Cavender-Bares, P. M. Groffman, M. Grove, J. B. Heffernan, S. E. Hobbie, J. Learned, J. L. Morse, C. Neill, L. A. Ogden, J. O'Neil-Dunne, D. E. Pataki, C. Polsky, R. Roy Chowdhury, M. Steele, and T. L. E. Trammell. 2016. Ecosystem services in managing residential landscapes: priorities, value dimensions, and cross-regional patterns. Urban Ecosystems 19:95–113.

LeBauer, D. S., and K. K. Treseder. 2008. Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. Ecology 89:371–379.

Leong, M., R. R. Dunn, and M. D. Trautwein. 2018. Biodiversity and socioeconomics in the city: a review of the luxury effect. Biology Letters 14:20180082.

Leveau, L. M., F. I. Isla, and M. I. Bellocq. 2015. Urbanization and the temporal homogenization of bird communities: a case study in central Argentina. Urban Ecosystems 18:1461–1476.

Leveau, L. M., and C. M. Leveau. 2016. Does urbanization affect the seasonal dynamics of bird communities in urban parks? Urban Ecosystems 19:631–647.

Levine, D. S., and M. J. Strube. 2012. Environmental attitudes, knowledge, intentions and behaviors among college students. Journal of Social Psychology 152:308–326.

Li, C., J. Li, and J. Wu. 2013. Quantifying the speed, growth modes, and landscape pattern changes of urbanization: a hierarchical patch dynamics approach. Landscape Ecology 28:1875–1888.

Liu, X., Y. Zhang, W. Han, A. Tang, J. Shen, Z. Cui, P. Vitousek, J. W. Erisman, K. Goulding, P. Christie, A. Fangmeier, and F. Zhang. 2013. Enhanced nitrogen deposition over China. Nature 494:459–462.

Locke, D. H., M. Avolio, T. Trammel, R. Roy Chowdhury, J. M. Grove, J. Rogan, D. G. Martin, N. Bettez, J. Cavender-Bares, P. M. Groffman, S. J. Hall, J. B. Heffernan, S. E. Hobbie, K. L. Larson, J. L. Morse, C. Neill, L. A. Ogden, J. P. M. O'Neil-Dunne,

D. Pataki, W. D. Pearse, C. Polsky, and M. M. Wheeler. 2018. A multi-city comparison of front and backyard differences in plant species diversity and nitrogen cycling in residential landscapes. Landscape and Urban Planning 178:102–111.

Lockett, L., T. Montague, C. McKenney, and D. Auld. 2002. Assessing public opinion on water conservation and water conserving landscapes in the semiarid southwestern United States. HortTechnology 12:392–396.

Loram, A., J. Tratalos, P. H. Warren, and K. J. Gaston. 2007. Urban domestic gardens (X): the extent & structure of the resource in five major cities. Landscape Ecology 22:601–615.

Lososová, Z., M. Chytrý, J. Danihelka, L. Tichý, and C. Ricotta. 2016. Biotic homogenization of urban floras by alien species: the role of species turnover and richness differences. Journal of Vegetation Science 27:452–459.

Lososová, Z., M. Chytrý, L. Tichý, J. Danihelka, K. Fajmon, O. Hájek, K. Kintrová, D. Láníková, Z. Otýpková, and V. Řehořek. 2012. Biotic homogenization of Central European urban floras depends on residence time of alien species and habitat types. Biological Conservation 145:179–184.

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

Lu, J. W. T., E. S. Svendsen, L. K. Campbell, J. Greenfeld, J. Braden, K. L. King, and N. Falxa-Raymond. 2010. Biological, social, and urban design factors affecting young street tree mortality in New York City. Cities and the Environment 3:5.

Luck, G. W., and L. T. Smallbone. 2011. The impact of urbanization on taxonomic and functional similarity among bird communities. Journal of Biogeography 38:894–906.

Luiselli, L., F. M. Angelici, and G. C. Akani. 2002. Comparative feeding strategies and dietary plasticity of the sympatric cobras Naja melanoleuca and Naja nigricollis in three diverging Afrotropical habitats. Canadian Journal of Zoology 80:55–63.

Mackay, C. M. L., and M. T. Schmitt. 2019. Do people who feel connected to nature do more to protect it? A meta-analysis. Journal of Environmental Psychology 65:101323.

Magallón, S., S. Gómez-Acevedo, L. L. Sánchez-Reyes, and T. Hernández-Hernández. 2015. A metacalibrated time-tree documents the early rise of flowering plant phylogenetic diversity. New Phytologist 207:437–453.

Magura, T., G. L. Lövei, and B. Tóthmérész. 2010. Does urbanization decrease diversity in ground beetle (Carabidae) assemblages? Global Ecology and Biogeography 19:16–26.

Marchetti, M. P., J. L. Lockwood, and T. Light. 2006. Effects of urbanization on California's fish diversity: differentiation, homogenization and the influence of spatial scale. Biological Conservation 127:310–318.

Martin, C. A. 2008. Landscape sustainability in a Sonoran Desert city. Cities and the Environment 1:5.

Martin, C. A., K. A. Peterson, and L. B. Stabler. 2003. Residential landscaping in Phoenix, Arizona, U.S.: practices and preferences relative to covenants, codes, and restrictions. Journal of Arboriculture 29:9–17.

Martini, N. F., and K. C. Nelson. 2015. The role of knowledge in residential lawn management. Urban Ecosystems 18:1031–1047.

Mata, L., C. E. Ramalho, J. Kennedy, K. M. Parris, L. Valentine, M. Miller, S. Bekessy, S. Hurley, and Z. Cumpston. 2020. Bringing nature back into cities. People and Nature 2:350–368.

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

McDonald, R. I., R. T. T. Forman, P. Kareiva, R. Neugarten, D. Salzer, and J. Fisher. 2009. Urban effects, distance, and protected areas in an urbanizing world. Landscape and Urban Planning 93:63–75.

McIntire, E. J. B., and A. Fajardo. 2014. Facilitation as a ubiquitous driver of biodiversity. New Phytologist 201:403–416.

McKinney, M. L. 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127:247–260.

McKinney, M. L. 2008. Effects of urbanization on species richness: a review of plants and animals. Urban Ecosystems 11:161–176.

McKinney, M. L., and J. L. Lockwood. 1999. Biotic homogenization: a few winners replacing many losers in the next mass extinction. Trends in Ecology & Evolution 14:450–453.

Miles, L. S., J. C. Johnson, R. J. Dyer, and B. C. Verrelli. 2018. Urbanization as a facilitator of gene flow in a human health pest. Molecular Ecology 27:3219–3230.

Møller, A. P. 2010. Interspecific variation in fear responses predicts urbanization in birds. Behavioral Ecology 21:365–371.

Morelli, F., Y. Benedetti, J. D. Ibáñez-Álamo, J. Jokimäki, R. Mänd, P. Tryjanowski, and A. P. Møller. 2016. Evidence of evolutionary homogenization of bird communities in urban environments across Europe. Global Ecology and Biogeography 25:1284–1293.

Moriuchi, K. S., D. L. Venable, C. E. Pake, and T. Lange. 2000. Direct measurement of the seed bank age structure of a Sonoran Desert annual plant. Ecology 81:1133–1138.

Mulroy, T. W., and P. W. Rundel. 1977. Annual plants: adaptations to desert environments. BioScience 27:109–114.

Murthy, A. C., T. S. Fristoe, and J. R. Burger. 2016. Homogenizing effects of cities on North American winter bird diversity. Ecosphere 7:e01216.

Namin, S., W. Xu, Y. Zhou, and K. Beyer. 2020. The legacy of the Home Owners' Loan Corporation and the political ecology of urban trees and air pollution in the United States. Social Science & Medicine 246:1112758.

Narango, D. L., D. W. Tallamy, and P. P. Marra. 2018. Nonnative plants reduce population growth of an insectivorous bird. Proceedings of the National Academy of Sciences USA 115:11549–11554.

Nassauer, J. I., Z. Wang, and E. Dayrell. 2009. What will the neighbors think? Cultural norms and ecological design. Landscape and Urban Planning 92:282–292.

Nelson, R. K., L. Winling, R. Marciano, and N. Connolly. 2020. Mapping inequality. American Panorama. https://dsl.richmond.edu/panorama/redlining/#loc=12/33.463/-112.152&city=phoenix-az.

Nesbitt, L., N. Hotte, S. Barron, J. Cowan, and S. R. J. Sheppard. 2017. The social and economic value of cultural ecosystem services provided by urban forests in North America: a review and suggestions for future research. Urban Forestry & Urban Greening 25:103–111.

Newbold, T., L. N. Hudson, S. Contu, S. L. L. Hill, J. Beck, Y. Liu, C. Meyer, H. R. P. Phillips, J. P. W. Scharlemann, and A. Purvis. 2018. Widespread winners and narrow-ranged losers: land use homogenizes biodiversity in local assemblages worldwide. PLoS Biology 16:e2006841.

Newbold, T., L. N. Hudson, S. L. L. Hill, S. Contu, I. Lysenko, R. A. Senior, L. Börger, D. J. Bennett, A. Choimes, B. Collen, J. Day, A. De Palma, S. Díaz, S. Echeverria-

Londoño, M. J. Edgar, A. Feldman, M. Garon, M. L. K. Harrison, T. Alhusseini, D. J. Ingram, Y. Itescu, J. Kattge, V. Kemp, L. Kirkpatrick, M. Kleyer, D. Laginha Pinto Correia, C. D. Martin, S. Meiri, M. Novosolov, Y. Pan, H. R. P. Phillips, D. W. Purves, A. Robinson, J. Simpson, S. L. Tuck, E. Weiher, H. J. White, R. M. Ewers, G. M. Mace, J. P. W. Scharlemann, and A. Purvis. 2015. Global effects of land use on local terrestrial biodiversity. Nature 520:45–50.

NOAA. 2018. Climate data online. National Climate Data Center. https://www.ncdc.noaa.gov/cdo-web/.

NOAA. 2019a. Extreme temperature facts for Phoenix and Yuma. National Weather Service. https://www.wrh.noaa.gov/psr/climate/extremeTemps.php.

NOAA. 2019b. Data tools: 1981-2010 normals. National Climate Data Center. https://www.ncdc.noaa.gov/cdo-web/datatools/normals.

Nock, C. A., A. Paquette, M. Follett, D. J. Nowak, and C. Messier. 2013. Effects of urbanization on tree species functional diversity in eastern North America. Ecosystems 16:1487–1497.

Nowak, D. J. 2012. Contrasting natural regeneration and tree planting in fourteen North American cities. Urban Forestry & Urban Greening 11:374–382.

Nowak, D. J., and E. J. Greenfield. 2012. Tree and impervious cover change in U.S. cities. Urban Forestry & Urban Greening 11:21–30.

Nowak, D. J., and E. J. Greenfield. 2018. Declining urban and community tree cover in the United States. Urban Forestry & Urban Greening 32:32–55.

Nowak, D. J., and J. T. Walton. 2005. Projected urban growth (2000-2050) and its estimated impact on the US forest resource. Journal of Forestry 103:383–389.

Noy-Meir, I. 1973. Desert ecosystems: environment and producers. Annual Review of Ecology and Systematics 4:25–51.

Öckinger, E., Å. Dannestam, and H. G. Smith. 2009. The importance of fragmentation and habitat quality of urban grasslands for butterfly diversity. Landscape and Urban Planning 93:31–37.

Oksanen, J., F. G. Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. H. H. Stevens, E. Szoecs, and H. Wagner. 2018. vegan: community ecology package. R package version 2.5-3. https://cran.r-project.org/package=vegan

Olden, J. D. 2006. Biotic homogenization: a new research agenda for conservation biogeography. Journal of Biogeography 33:2027–2039.

Olden, J. D., L. Comte, and X. Giam. 2018. The Homogocene: a research prospectus for the study of biotic homogenisation. NeoBiota 37:23–36.

Oliveira, J. P., I. Sousa-Pinto, G. M. Weber, and I. Bertocci. 2014. Urban vs. extra-urban environments: scales of variation of intertidal benthic assemblages in north Portugal. Marine Environmental Research 97:48–57.

Ossola, A., D. Locke, B. Lin, and E. Minor. 2019. Greening in style: urban form, architecture and the structure of front and backyard vegetation. Landscape and Urban Planning 185:141–157.

Padullés Cubino, J., M. L. Avolio, M. M. Wheeler, K. L. Larson, S. E. Hobbie, J. Cavender-Bares, S. J. Hall, K. C. Nelson, T. L. E. Trammell, C. Neill, D. E. Pataki, J. M. Grove, and P. M. Groffman. 2020. Linking yard plant diversity to homeowners' landscaping priorities across the U.S. Landscape and Urban Planning 196:103730.

Padullés Cubino, J., J. Cavender-Bares, S. E. Hobbie, S. J. Hall, T. L. E. Trammell, C. Neill, M. L. Avolio, L. E. Darling, and P. M. Groffman. 2019. Contribution of non-native plants to the phylogenetic homogenization of U.S. yard floras. Ecosphere 10:e02638.

Padullés Cubino, J., J. Cavender-Bares, S. E. Hobbie, D. E. Pataki, M. L. Avolio, L. E. Darling, K. L. Larson, S. J. Hall, P. M. Groffman, T. L. E. Trammell, M. K. Steele, J. M. Grove, and C. Neill. 2018. Drivers of plant species richness and phylogenetic composition in urban yards at the continental scale. Landscape Ecology 34:63–77.

Pake, C. E., and D. L. Venable. 1995. Is coexistence of Sonoran Desert annuals mediated by temporal variability in reproductive success? Ecology 76:246–261.

Pardee, G. L., and S. M. Philpott. 2014. Native plants are the bee's knees: local and landscape predictors of bee richness and abundance in backyard gardens. Urban Ecosystems 17:641–659.

Pardo, L. H., M. E. Fenn, C. L. Goodale, L. H. Geiser, C. T. Driscoll, E. B. Allen, J. S. Baron, R. Bobbink, W. D. Bowman, C. M. Clark, B. Emmett, F. S. Gilliam, T. L. Greaver, S. J. Hall, E. A. Lilleskov, L. Liu, J. A. Lynch, K. J. Nadelhoffer, S. S. Perakis, M. J. Robin-Abbott, J. L. Stoddard, K. C. Weathers, and R. L. Dennis. 2011. Effects of nitrogen deposition and empirical nitrogen critical loads for ecoregions of the United States. Ecological Applications 21:3049–3082.

Pataki, D. E. 2015. Grand challenges in urban ecology. Frontiers in Ecology and Evolution 3:57.

Payne, R. J., N. B. Dise, C. D. Field, A. J. Dore, S. J. M. Caporn, and C. J. Stevens. 2017. Nitrogen deposition and plant biodiversity: past, present, and future. Frontiers in Ecology and the Environment 15:431–436.

Pearsall, H., and Z. Christman. 2012. Tree-lined lanes or vacant lots? Evaluating non-stationarity between urban greenness and socio-economic conditions in Philadelphia, Pennsylvania, USA at multiple scales. Applied Geography 35:257–264.

Pearse, W. D., M. W. Cadotte, J. Cavender-Bares, A. R. Ives, C. Tucker, S. C. Walker, and M. R. Helmus. 2015. pez: phylogenetics for the environmental sciences. Bioinformatics 31:2888–2890.

Pearse, W. D., J. Cavender-Bares, S. E. Hobbie, M. L. Avolio, N. Bettez, R. Roy Chowdhury, L. E. Darling, P. M. Groffman, J. M. Grove, S. J. Hall, J. B. Heffernan, J. Learned, C. Neill, K. C. Nelson, D. E. Pataki, B. L. Ruddell, M. E. Steele, and T. L. E. Trammell. 2018. Ecological homogenisation in North American urban yards: vegetation diversity, composition, and structure. Ecosphere 9:e02105.

Peterson, M. N., B. Thurmond, M. McHale, S. Rodriguez, H. D. Bondell, and M. Cook. 2012. Predicting native plant landscaping preferences in urban areas. Sustainable Cities and Society 5:70–76.

Phoenix, Arizona, Community Forest Assessment. 2014. Davey Resource Group, Phoenix, Arizona, USA.

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

Pickett, S. T. A., M. L. Cadenasso, J. M. Grove, P. M. Groffman, L. E. Band, C. G. Boone, W. R. Burch, C. S. B. Grimmond, J. Hom, J. C. Jenkins, N. L. Law, C. H. Nilon, R. V. Pouyat, K. Szlavecz, P. S. Warren, and M. A. Wilson. 2008. Beyond urban legends: an emerging framework of urban ecology, as illustrated by the Baltimore Ecosystem Study. BioScience 58:139–150.

Pickett, S. T. A., M. L. Cadenasso, and S. J. Meiners. 2009. Ever since Clements: from succession to vegetation dynamics and understanding to intervention. Applied Vegetation Science 12:9–21.

Pincetl, S., T. W. Gillespie, D. E. Pataki, E. Porse, S. Jia, E. Kidera, N. Nobles, J. Rodriguez, and D. Choi. 2019. Evaluating the effects of turf-replacement programs in Los Angeles. Landscape and Urban Planning 185:210–221.

Pino, J., X. Font, M. de Cáceres, and R. Molowny-Horas. 2009. Floristic homogenization by native ruderal and alien plants in north-east Spain: the effect of environmental differences on a regional scale. Global Ecology and Biogeography 18:563–574.

Polakowski, N. R., V. I. Lohr, and T. Cerny-Koenig. 2011. Survey of wholesale production nurseries indicates need for more education on the importance of plant species diversity. Arboriculture & Urban Forestry 37:259–264.

Polsky, C., J. M. Grove, C. Knudson, P. M. Groffman, N. Bettez, J. Cavender-Bares, S. J. Hall, J. B. Heffernan, S. E. Hobbie, K. L. Larson, J. L. Morse, C. Neill, K. C. Nelson, L. A. Ogden, J. O'Neil-Dunne, D. E. Pataki, R. Roy Chowdhury, and M. K. Steele. 2014. Assessing the homogenization of urban land management with an application to US residential lawn care. Proceedings of the National Academy of Sciences USA 111:4432–4437.

Pool, T. K., and J. D. Olden. 2012. Taxonomic and functional homogenization of an endemic desert fish fauna. Diversity and Distributions 18:366–376.

Potts, L. E., M. J. Roll, and S. J. Wallner. 2002. Colorado native plant survey — voices of the green industry. Native Plants Journal 3:121–125.

Potvin, D. A., and K. M. Parris. 2012. Song convergence in multiple urban populations of silvereyes (*Zosterops lateralis*). Ecology and Evolution 2:1977–1984.

Pouyat, R. V., I. D. Yesilonis, M. Dombos, K. Szlavecz, H. Setälä, S. Cilliers, E. Hornung, D. J. Kotze, and S. Yarwood. 2015. A global comparison of surface soil characteristics across five cities: a test of the urban ecosystem convergence hypothesis. Soil Science 180:136–145.

Pulsford, S. A., D. B. Lindenmayer, and D. A. Driscoll. 2016. A succession of theories: purging redundancy from disturbance theory. Biological Reviews 91:148–167.

R Core Team. 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rahel, F. J. 2000. Homogenization of fish faunas across the United States. Science 288:854–856.

Ramage, B. S., L. A. Roman, and J. S. Dukes. 2013. Relationships between urban tree communities and the biomes in which they reside. Applied Vegetation Science 16:8–20.

Ramalho, C. E., and R. J. Hobbs. 2012. Time for a change: dynamic urban ecology. Trends in Ecology & Evolution 27:179–188.

Rao, L. E., and E. B. Allen. 2010. Combined effects of precipitation and nitrogen deposition on native and invasive winter annual production in California deserts. Oecologia 162:1035–1046.

Raudsepp-Hearne, C., G. D. Peterson, E. M. Bennett, R. Biggs, A. V. Norström, L. Pereira, J. Vervoort, D. M. Iwaniec, T. McPhearson, P. Olsson, T. Hichert, M. Falardeau, and A. Jiménez Aceituno. 2019. Seeds of good anthropocenes: developing sustainability scenarios for Northern Europe. Sustainability Science 15:605–617.

Reich, P. B. 2014. The world-wide "fast-slow" plant economics spectrum: a traits manifesto. Journal of Ecology 102:275–301.

Reichert, B. E., A. R. Sovie, B. J. Udell, K. M. Hart, R. R. Borkhataria, M. Bonneau, R. Reed, and R. McCleery. 2017. Urbanization may limit impacts of an invasive predator on native mammal diversity. Diversity and Distributions 23:355–367.

Ricotta, C., L. Celesti-Grapow, I. Kühn, G. Rapson, P. Pyšek, F. A. La Sorte, and K. Thompson. 2014. Geographical constraints are stronger than invasion patterns for European urban floras. PLoS ONE 9:e85661.

Ricotta, C., G. L. Rapson, U. Asmus, P. Pyšek, I. Kühn, F. A. La Sorte, and K. Thompson. 2017. British plants as aliens in New Zealand cities: residence time moderates their impact on the beta diversity of urban floras. Biological Invasions 19:3589–3599.

Ripley, B., B. Venables, D. M. Bates, K. Hornik, A. Gebhardt, and D. Firth. 2019. Support Functions and Datasets for Venables and Ripley's MASS. R package version 7.3-51.4. https://CRAN.R-project.org/package=MASS

Ripplinger, J., S. L. Collins, A. M. York, and J. Franklin. 2017. Boom-bust economics and vegetation dynamics in a desert city: how strong is the link? Ecosphere 8:e01826.

Ripplinger, J., J. Franklin, and S. L. Collins. 2016. When the economic engine stalls - a multi-scale comparison of vegetation dynamics in pre- and post-recession Phoenix, Arizona, USA. Landscape and Urban Planning 153:140–148.

Robbins, P. 2007. Lawn people: how grasses, weeds, and chemicals make us who we are. Temple University Press, Philadelphia, Pennsylvania, USA.

Robinson, S. L., and J. T. Lundholm. 2012. Ecosystem services provided by urban spontaneous vegetation. Urban Ecosystems 15:545–557.

Rocha, A. M., A. González-Reyes, J. Corronca, S. Rodríguez-Artigas, I. Doma, E. Y. Repp, and X. Acosta. 2016. Tardigrade diversity: an evaluation of natural and disturbed environments of the province of Salta (Argentina). Zoological Journal of the Linnean Society 178:755–764.

Rodrigues, J. L. M., V. H. Pellizari, R. Mueller, K. Baek, E. da C. Jesus, F. S. Paula, B. Mirza, G. S. Hamaou, S. M. Tsai, B. Feiglf, J. M. Tiedje, B. J. M. Bohannan, and K. Nüsslein. 2013. Conversion of the Amazon rainforest to agriculture results in biotic homogenization of soil bacterial communities. Proceedings of the National Academy of Sciences USA 110:988–993.

Rogers, D. A., T. P. Rooney, T. J. Hawbaker, V. C. Radeloff, and D. M. Waller. 2009. Paying the extinction debt in southern Wisconsin forest understories. Conservation Biology 23:1497–1506.

Roman, L. A., J. J. Battles, and J. R. McBride. 2014. Determinants of establishment survival for residential trees in Sacramento County, CA. Landscape and Urban Planning 129:22–31.

Roman, L. A., J. P. Fristensky, T. S. Eisenman, E. J. Greenfield, R. E. Lundgren, C. E. Cerwinka, D. A. Hewitt, and C. C. Welsh. 2017. Growing canopy on a college campus: understanding urban forest change through archival records and aerial photography. Environmental Management 60:1042–1061.

Roman, L. A., H. Pearsall, T. S. Eisenman, T. M. Conway, R. T. Fahey, S. Landry, J. Vogt, N. S. van Doorn, J. M. Grove, D. H. Locke, A. C. Bardekjian, J. J. Battles, M. L. Cadenasso, C. C. K. van den Bosch, M. Avolio, A. Berland, G. D. Jenerette, S. K. Mincey, D. E. Pataki, and C. Staudhammer. 2018. Human and biophysical legacies shape contemporary urban forests: a literature synthesis. Urban Forestry & Urban Greening 31:157–168.

Roman, L. A., and F. N. Scatena. 2011. Street tree survival rates: meta-analysis of previous studies and application to a field survey in Philadelphia, PA, USA. Urban Forestry & Urban Greening 10:269–274.

Roman, L. A., L. A. Walker, C. M. Martineau, D. J. Muffly, S. A. MacQueen, and W. Harris. 2015. Stewardship matters: case studies in establishment success of urban trees. Urban Forestry & Urban Greening 14:1174–1182.

Rooney, T. P., J. D. Olden, M. K. Leach, and D. A. Rogers. 2007. Biotic homogenization and conservation prioritization. Biological Conservation 134:447–450.

Rosseel, Y. 2012. lavaan: An R package for structural equation modeling. Journal of Statistical Software 48:1–36.

Ryan, J. 2011. Anthoethnography: emerging research into the culture of flora, aesthetic experience of plants, and the wildflower tourism of the future. New Scholar 1:28–40.

Sala, O. E., F. S. III Chapin, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, M. Oesterheld, N. L. Poff, M. T. Skykes, B. H. Walker, M. Walker, and D. H. Wall. 2000. Global biodiversity scenarios for the year 2100. Science 287:1770–1774.

Samani, Z. A., and M. Pessarakli. 1986. Estimating potential crop evapotranspiration with minimum data in Arizona. Transactions of the ASAE 29:522–524.

Schade, J. D., and S. E. Hobbie. 2005. Spatial and temporal variation in islands of fertility in the Sonoran Desert. Biogeochemistry 73:541–553.

Scharenbroch, B. C., J. E. Lloyd, and J. L. Johnson-Maynard. 2005. Distinguishing urban soils with physical, chemical, and biological properties. Pedobiologia 49:283–296.

Schlesinger, W. H., and A. M. Pilmanis. 1998. Plant-soil interactions in deserts. Biogeochemistry 42:169–187.

Schlesinger, W. H., J. A. Raikes, A. E. Hartley, and A. F. Cross. 1996. On the spatial pattern of soil nutrients in desert ecosystems. Ecology 77:364–374.

Schneider, A., M. A. Friedl, and D. Potere. 2009. A new map of global urban extent from MODIS satellite data. Environmental Research Letters 4:044003.

Schroeder, P. J., and D. G. Jenkins. 2018. How robust are popular beta diversity indices to sampling error? Ecosphere 9:e02100.

Schultz, P. W. 2011. Conservation means behavior. Conservation Biology 25:1080–1083.

Scott, M. C. 2006. Winners and losers among stream fishes in relation to land use legacies and urban development in the southeastern US. Biological Conservation 127:301–309.

Seto, K. C., M. Fragkias, B. Güneralp, and M. K. Reilly. 2011. A meta-analysis of global urban land expansion. PLoS ONE 6:e23777.

Sisser, J. M., K. C. Nelson, K. L. Larson, L. A. Ogden, C. Polsky, and R. Roy Chowdhury. 2016. Lawn enforcement: how municipal policies and neighborhood norms influence homeowner residential landscape management. Landscape and Urban Planning 150:16–25.

Smith, I. A., V. K. Dearborn, and L. R. Hutyra. 2019. Live fast, die young: accelerated growth, mortality, and turnover in urban street trees. PLoS ONE 14:e0215846.

Smith, S. A., and J. W. Brown. 2018. Constructing a broadly inclusive seed plant phylogeny. American Journal of Botany 105:302–314.

Snyman, H. A. 2002. Short-term response of rangeland botanical composition and productivity to fertilization (N and P) in a semi-arid climate of South Africa. Journal of Arid Environments 50:167–183.

Soanes, K., M. Sievers, Y. E. Chee, N. S. G. Williams, M. Bhardwaj, A. J. Marshall, and K. M. Parris. 2019. Correcting common misconceptions to inspire conservation action in urban environments. Conservation Biology 33:300–306.

Soga, M., and K. J. Gaston. 2016. Extinction of experience: the loss of human-nature interactions. Frontiers in Ecology and the Environment 14:94–101.

Sokol, E. R., B. L. Brown, C. C. Carey, B. M. Tornwall, C. M. Swan, and J. E. Barrett. 2015. Linking management to biodiversity in built ponds using metacommunity simulations. Ecological Modelling 296:36–45.

Sol, D., C. González-Lagos, D. Moreira, J. Maspons, and O. Lapiedra. 2014. Urbanisation tolerance and the loss of avian diversity. Ecology Letters 17:942–950.

Spinti, J. E., R. St. Hilaire, and D. VanLeeuwen. 2004. Balancing landscape preferences and water conservation in a desert community. HortTechnology 14:72–77.

Sponseller, R. A., S. J. Hall, D. P. Huber, N. B. Grimm, J. P. Kaye, C. M. Clark, and S. L. Collins. 2012. Variation in monsoon precipitation drives spatial and temporal patterns of *Larrea tridentata* growth in the Sonoran Desert. Functional Ecology 26:750–758.

St. Hilaire, R., D. M. VanLeeuwen, and P. Torres. 2010. Landscape preferences and water conservation choices of residents in a high desert environment. HortTechnology 20:308–314.

Steele, M. K., J. B. Heffernan, N. Bettez, J. Cavender-Bares, P. M. Groffman, J. M. Grove, S. Hall, S. E. Hobbie, K. Larson, J. L. Morse, C. Neill, K. C. Nelson, J. O'Neil-Dunne, L. Ogden, D. E. Pataki, C. Polsky, and R. Roy Chowdhury. 2014. Convergent surface water distributions in U.S. cities. Ecosystems 17:685–697.

Taasoobshirazi, G., and S. Wang. 2016. The performance of the SRMR, RMSEA, CFI, AND TLI: an examination of sample size, path size, and degrees of freedom. Journal of Applied Quantitative Methods 11:31–41.

Taylor, L., and D. F. Hochuli. 2015. Creating better cities: how biodiversity and ecosystem functioning enhance urban residents' wellbeing. Urban Ecosystems 18:747–762.

The Plant List. 2013. Version 1.1. http://www.theplantlist.org/.

Thompson, K., and M. A. McCarthy. 2008. Traits of British alien and native urban plants. Journal of Ecology 96:853–859.

Threlfall, C. G., and D. Kendal. 2018. The distinct ecological and social roles that wild spaces play in urban ecosystems. Urban Forestry & Urban Greening 29:348–356.

Threlfall, C. G., L. Mata, J. A. Mackie, A. K. Hahs, N. E. Stork, N. S. G. Williams, and S. J. Livesley. 2017. Increasing biodiversity in urban green spaces through simple vegetation interventions. Journal of Applied Ecology 54:1874–1883.

Trentanovi, G., M. von der Lippe, T. Sitzia, U. Ziechmann, I. Kowarik, and A. Cierjacks. 2013. Biotic homogenization at the community scale: disentangling the roles of urbanization and plant invasion. Diversity and Distributions 19:738–748.

Tubby, K. V., and J. F. Webber. 2010. Pests and diseases threatening urban trees under a changing climate. Forestry 83:451–459.

Tucker, C. M., M. W. Cadotte, S. B. Carvalho, T. J. Davies, S. Ferrier, S. A. Fritz, R. Grenyer, M. R. Helmus, L. S. Jin, A. O. Mooers, S. Pavoine, O. Purschke, D. W. Redding, D. F. Rosauer, M. Winter, and F. Mazel. 2017. A guide to phylogenetic metrics for conservation, community ecology and macroecology. Biological Reviews 92:698–715.

Turner, V. K., and M. Stiller. 2020. How do homeowners associations regulate residential landscapes? An analysis of rule structure and content in Maricopa County (AZ). Journal of the American Planning Association 86:25–38.

Turner, W. R., T. Nakamura, and M. Dinetti. 2004. Global urbanization and the separation of humans from nature. BioScience 54:585.

U.S. Census Bureau. 2019a. New Census Bureau estimates show counties in South and West lead nation in population growth. https://www.census.gov/newsroom/press-releases/2019/estimates-county-metro.html.

U.S. Census Bureau. 2019b. Highlights of annual 2019 characteristics of new housing. https://www.census.gov/construction/chars/highlights.html.

Uchida, K., H. Fujimoto, and A. Ushimaru. 2018. Urbanization promotes the loss of seasonal dynamics in the semi-natural grasslands of an East Asian megacity. Basic and Applied Ecology 29:1–11.

Ulrich, W., M. Zalewski, and K. Komosiński. 2007. Diversity of carrion visiting beetles at rural and urban sites. Community Ecology 8:171–181.

UNEP. 1992. World Atlas of Desertification. Edward Arnold, New York, New York, USA.

United Nations. 2014. World Urbanization Prospects: The 2014 Revision.

Uren, H. V., P. L. Dzidic, and B. J. Bishop. 2015. Exploring social and cultural norms to promote ecologically sensitive residential garden design. Landscape and Urban Planning 137:76–84.

USDA NRCS. 2020. The PLANTS database. National Plant Data Team, Greensboro, North Carolina, USA. http://plants.usda.gov.

van Heezik, Y. M., K. J. M. Dickinson, and C. Freeman. 2012. Closing the gap: communicating to change gardening practices in support of native biodiversity in urban private gardens. Ecology and Society 17:34.

van Heezik, Y., C. Freeman, K. Davidson, and B. Lewis. 2020. Uptake and engagement of activities to promote native species in private gardens. Environmental Management 66:42–55.

Vangestel, C., J. Mergeay, D. A. Dawson, V. Vandomme, and L. Lens. 2011. Spatial heterogeneity in genetic relatedness among house sparrows along an urban–rural gradient as revealed by individual-based analysis. Molecular Ecology 20:4643–4653.

Vázquez-Reyes, L. D., M. del C. Arizmendi, H. O. Godínez-Álvarez, and A. G. Navarro-Sigüenza. 2017. Directional effects of biotic homogenization of bird communities in Mexican seasonal forests. The Condor 119:275–288.

Vellend, M., K. Verheyen, K. M. Flinn, H. Jacquemyn, A. Kolb, H. Van Calster, G. Peterken, B. J. Graae, J. Bellemare, O. Honnay, J. Brunet, M. Wulf, F. Gerhardt, and M. Hermy. 2007. Homogenization of forest plant communities and weakening of species-environment relationships via agricultural land use. Journal of Ecology 95:565–573.

Venable, D. L. 2007. Bet hedging in a guild of desert annuals. Ecology 88:1086–1090.

Venable, D. L., and C. E. Pake. 1999. Population ecology of Sonoran Desert annual plants. Pages 115–142 *in* R. H. Robichaux, editor. Ecology of Sonoran Desert Plants and Plant Communities. University of Arizona Press.

Venable, D. L., C. E. Pake, and A. C. Caprio. 1993. Diversity and coexistence of Sonoran Desert winter annuals. Plant Species Biology 8:207–216.

Villaseñor, J. L. 2016. Checklist of the native vascular plants of Mexico. Revista Mexicana de Biodiversidad 87:559–902.

Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997. Human domination of Earth's ecosystems. Science 277:494–499.

Vogt, J. M., S. L. Watkins, S. K. Mincey, M. S. Patterson, and B. C. Fischer. 2015. Explaining planted-tree survival and growth in urban neighborhoods: a social-ecological approach to studying recently-planted trees in Indianapolis. Landscape and Urban Planning 136:130–143.

von der Lippe, M., and I. Kowarik. 2008. Do cities export biodiversity? Traffic as dispersal vector across urban-rural gradients. Diversity and Distributions 14:18–25.

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

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgon. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706–723.

Wang, G., J. Zuo, X.-R. Li, Y. Liu, J. Yu, H. Shao, and Y. Li. 2014. Low plant diversity and floristic homogenization in fast-urbanizing towns in Shandong Peninsular, China: effects of urban greening at regional scale for ecological engineering. Ecological Engineering 64:179–185.

Webb, C. O., D. D. Ackerly, M. A. McPeek, and M. J. Donoghue. 2002. Phylogenies and community ecology. Annual Review of Ecology and Systematics 33:475–505.

Wheeler, M. M., K. L. Larson, and R. Andrade. 2020. Attitudinal and structural drivers of preferred versus actual residential landscapes in a desert city. Urban Ecosystems 23:659–673.

White, A., J. B. Fant, K. Havens, M. Skinner, and A. T. Kramer. 2018. Restoring species diversity: assessing capacity in the U.S. native plant industry. Restoration Ecology 26:605–611.

Widney, S., B. C. Fischer, and J. Vogt. 2016. Tree mortality undercuts ability of tree-planting programs to provide benefits: results of a three-city study. Forests 7:65.

Wilke, A. B. B., G. C. de Carvalho, and M. T. Marrelli. 2018. Retention of ancestral polymorphism in *Culex nigripalpus* (Diptera: Culicidae) from São Paulo, Brazil. Infection, Genetics and Evolution 65:333–339.

Williams, N. S. G., A. K. Hahs, and P. A. Vesk. 2015. Urbanisation, plant traits and the composition of urban floras. Perspectives in Plant Ecology, Evolution and Systematics 17:78–86.

Wu, J., G. D. Jenerette, A. Buyantuyev, and C. L. Redman. 2011. Quantifying spatiotemporal patterns of urbanization: the case of the two fastest growing metropolitan regions in the United States. Ecological Complexity 8:1–8.

Yahdjian, L., L. Gherardi, and O. E. Sala. 2011. Nitrogen limitation in arid-subhumid ecosystems: a meta-analysis of fertilization studies. Journal of Arid Environments 75:675–680.

Yang, J., F. A. La Sorte, P. Pyšek, P. Yan, D. Nowak, and J. McBride. 2015. The compositional similarity of urban forests among the world's cities is scale dependent. Global Ecology and Biogeography 24:1413–1423.

Yu, G., D. Smith, H. Zhu, Y. Guan, and T. T.-Y. Lam. 2017. ggtree: an R package for visualization and annotation of phylogenetic trees with their covariates and other associated data. Methods in Ecology and Evolution 8:28–36.

Yue, C., T. Hurley, and N. O. Anderson. 2012. Heterogeneous consumer preferences for native and invasive plants: evidence from experimental auctions. HortScience 47:1091–1095.

Zanne, A. E., D. C. Tank, W. K. Cornwell, J. M. Eastman, S. A. Smith, R. G. FitzJohn, D. J. McGlinn, B. C. O'Meara, A. T. Moles, P. B. Reich, D. L. Royer, D. E. Soltis, P. F. Stevens, M. Westoby, I. J. Wright, L. Aarssen, R. I. Bertin, A. Calaminus, R. Govaerts, F. Hemmings, M. R. Leishman, J. Oleksyn, P. S. Soltis, N. G. Swenson, L. Warman, and J. M. Beaulieu. 2014. Three keys to the radiation of angiosperms into freezing environments. Nature 506:89–92.

Ziter, C., R. A. Graves, and M. G. Turner. 2017. How do land-use legacies affect ecosystem services in United States cultural landscapes? Landscape Ecology 32:2205–2218.

Zmyslony, J., and D. Gagnon. 1998. Residential management of urban front-yard landscape: a random process? Landscape and Urban Planning 40:295–307.

Zube, E. H., D. E. Simcox, and C. S. Law. 1986. The oasis image in two desert cities. Landscape Research 11:7–11.

APPENDIX A

CHAPTER 2 SUPPLEMENTARY TABLES

TABLE S1. Journal titles for which all articles obtained by the original search were excluded.

Journal title	Number of papers
Accident Analysis and Prevention	14
ACM Computing Surveys	1
Advances in Nutrition	1
Advances in Space Research	4
Agricultural Economics-Zemedelska Ekonomika	1
AIDS	2
Aids Care-Psychological and Socio-Medical Aspects of AIDS/HIV	2
Air Quality Atmosphere and Health	7
Alcoholism-Clinical and Experimental Research	1
American Journal of Agricultural Economics	1
American Journal of Cancer Research	1
American Journal of Cardiology	2
American Journal of Clinical Nutrition	2
American Journal of Clinical Oncology-Cancer Clinical Trials	1
American Journal of Community Psychology	2
American Journal of Economics and Sociology	1
American Journal of Education	1
American Journal of Epidemiology	9
American Journal of Health Promotion	2
American Journal of Human Biology	3
American Journal of Preventive Medicine	3
American Journal of Public Health	11
American Journal of Respiratory and Critical Care Medicine	5
American Journal of Sociology	3
American Journal of Tropical Medicine and Hygiene	8
American Sociological Review	4
Analytic Methods in Accident Research	3
Annals of Agricultural and Environmental Medicine	2
Annals of Human Biology	1
Annals of Tropical Medicine and Parasitology	1
Anticancer Research	1
Antimicrobial Agents and Chemotherapy	3
Applied Economics	3
Applied Mathematics and Computation	1
Architectural Design	2
Archives of Environmental & Occupational Health	1

Journal title	Number of papers
Archives of Sexual Behavior	5
Archives of Toxicology	1
Arquivos De Neuro-Psiquiatria	2
Asia-Pacific Journal of Public Health	1
Asian Pacific Journal of Cancer Prevention	1
Atmospheric Measurement Techniques	9
Australian Psychologist	3
Automation in Construction	1
Bell Journal of Economics	1
Best Practice & Research in Clinical Rheumatology	1
Biological Psychiatry	1
Biological Trace Element Research	2
Biosecurity and Bioterrorism-Biodefense Strategy Practice and Science	2
BMC Health Services Research	3
BMC Infectious Diseases	3
BMC Medical Research Methodology	2
BMC Public Health	21
BMJ Global Health	1
Breast Cancer Research	1
British Journal of Sociology	1
Building Research and Information	2
Bulletin of Indonesian Economic Studies	1
Bulletin of The Seismological Society of America	2
Canadian Journal of Agricultural Economics-Revue Canadienne D Agroeconomie	1
Canadian Journal of Public Health-Revue Canadienne De Sante Publique	1
Canadian Journal on Aging-Revue Canadienne Du Vieillissement	2
Cancer	2
Cancer Causes & Control	4
Cancer Epidemiology Biomarkers & Prevention	1
Case Studies on Transport Policy	5
Central African Journal of Medicine	1
Chemical Engineering Journal	2
Chest	2
Child Maltreatment	1
China Economic Review	3
Clinical and Experimental Allergy	1
Clinical Cardiology	1
Clinical Infectious Diseases	1

Journal title	Number of papers
Clinical Neurology and Neurosurgery	1
Clinical Pediatric Emergency Medicine	2
Cluster Computing-The Journal of Networks Software Tools and Applications	1
Cogent Economics & Finance	1
Community Mental Health Journal	1
Computational Economics	1
Computer Communications	4
Computer Networks	3
Computers & Geosciences	1
Computers & Graphics-UK	1
Computers and Electronics in Agriculture	3
Computers Environment and Urban Systems	20
Computers in Industry	1
Concurrency and Computation-Practice & Experience	1
Crime and Justice-A Review of Research	1
Criminology	4
Culture Health & Sexuality	2
Demographic Research	2
Demography	3
Diabetes	1
Diabetic Medicine	1
Discourse-Studies in The Cultural Politics of Education	1
Drug and Alcohol Dependence	2
Ecohealth	3
Ecological Economics	40
Ecology of Food and Nutrition	1
Economic and Social Changes-Facts Trends Forecast	1
Economic Botany	1
Economic History Review	2
Economic Modelling	6
Economic Research-Ekonomska Istrazivanja	1
Economics of Transportation	1
Electronic Commerce Research and Applications	1
Emerging Infectious Diseases	2
Empirical Economics	1
Energy Economics	4
Energy Research & Social Science	4
Energy Sources Part B-Economics Planning and Policy	3

Journal title	Number of papers
Engineering Computations	1
Engineering Geology	2
Entrepreneurial Business and Economics Review	1
Environmental & Resource Economics	1
Environmental Fluid Mechanics	3
Environmental Geochemistry and Health	5
Environmental Health	3
Environmental Health Insights	1
Environmental Health Perspectives	27
Environmental Modelling & Software	10
Epidemiologic Reviews	1
Epidemiology	6
Eurasip Journal on Wireless Communications and Networking	3
European Journal of Cancer	1
European Journal of Clinical Microbiology & Infectious Diseases	1
European Journal of Marketing	2
European Psychiatry	1
European Transport Research Review	3
Family Planning Perspectives	3
Faraday Discussions	2
FEMS Yeast Research	1
Food Security	1
Forest Policy and Economics	5
Frontiers in Public Health	1
Future Generation Computer Systems-The International Journal of EScience	4
Future Generation Computer Systems-The International Journal of Grid Computing-Theory Methods and Applications	1
Gender Place and Culture	4
Geoscientific Instrumentation Methods and Data Systems	2
Geoscientific Model Development	5
Geospatial Health	4
Global Health Action	2
Global Health Promotion	1
Global Health-Science and Practice	1
Gut	1
Health & Place	9
Health and Quality of Life Outcomes	2
Health Communication	1
Health Economics	1

Journal title	Number of papers
Health Education & Behavior	2
Health Policy	2
Health Policy and Planning	1
Health Promotion International	4
Health Scope	1
Health Security	1
Health Services Research	2
Herd-Health Environments Research & Design Journal	1
Hitotsubashi Journal of Economics	1
Housing Studies	2
IEEE Access	14
IEEE Antennas and Wireless Propagation Letters	2
IEEE Communications Letters	3
IEEE Communications Magazine	3
IEEE Geoscience and Remote Sensing Letters	4
IEEE Geoscience and Remote Sensing Magazine	1
IEEE Intelligent Transportation Systems Magazine	2
IEEE Internet of Things Journal	4
IEEE Journal of Selected Topics in Applied Earth Observations and Remote Sensing	7
IEEE Journal on Selected Areas in Communications	3
IEEE Network	2
IEEE Robotics and Automation Letters	1
IEEE Sensors Journal	2
IEEE Transactions on Aerospace and Electronic Systems	1
IEEE Transactions on Antennas and Propagation	5
IEEE Transactions on Automation Science and Engineering	1
IEEE Transactions on Broadcasting	1
IEEE Transactions on Computers	1
IEEE Transactions on Evolutionary Computation	1
IEEE Transactions on Geoscience and Remote Sensing	2
IEEE Transactions on Industrial Electronics	3
IEEE Transactions on Industrial Informatics	1
IEEE Transactions on Intelligent Transportation Systems	9
IEEE Transactions on Mobile Computing	4
IEEE Transactions on Parallel and Distributed Systems	1
IEEE Transactions on Pattern Analysis and Machine Intelligence	1
IEEE Transactions on Power Electronics	1
IEEE Transactions on Robotics	2

Journal title	Number of papers
IEEE Transactions on Smart Grid	1
IEEE Transactions on Vehicular Technology	13
IEEE Transactions on Visualization and Computer Graphics	1
IEEE Transactions on Wireless Communications	6
IEEE-ACM Transactions on Networking	2
IEEE-CAA Journal of Automatica Sinica	1
IEICE Transactions on Communications	2
IEICE Transactions on Fundamentals of Electronics Communications and Computer Sciences	1
IET Intelligent Transport Systems	2
Impact Assessment and Project Appraisal	4
Indian Pediatrics	2
Industrial Health	1
Infectious Diseases of Poverty	2
Inflammatory Bowel Diseases	2
Inhalation Toxicology	4
International Affairs	1
International Journal for Quality in Health Care	1
International Journal of Advanced Computer Science and Applications	1
International Journal of Antennas and Propagation	2
International Journal of Behavioral Nutrition and Physical Activity	2
International Journal of Cancer	1
International Journal of Computer Science and Network Security	1
International Journal of Computer Vision	1
International Journal of Cultural Policy	1
International Journal of Digital Earth	5
International Journal of Distributed Sensor Networks	3
International Journal of E-Planning Research	2
International Journal of Environmental Health Research	2
International Journal of Environmental Research and Public Health	41
International Journal of Epidemiology	3
International Journal of Health Care Finance & Economics	1
International Journal of Health Geographics	14
International Journal of Heritage Studies	3
International Journal of Human Rights in Health Care	1
International Journal of Human-Computer Interaction	1
International Journal of Life Cycle Assessment	4
International Journal of Medical Research & Health Sciences	1
International Journal of Retail & Distribution Management	2

Journal title	Number of papers
International Journal of Robotics Research	2
International Journal of Rock Mechanics and Mining Sciences	1
International Journal of Semantic Computing	1
International Journal of Social Economics	3
International Journal of Sustainable Transportation	5
International Labor and Working-Class History	2
Isotopes in Environmental and Health Studies	1
Israel Journal of Health Policy Research	1
Jama Psychiatry	2
JASSS-The Journal of Artificial Societies and Social Simulation	3
Journal of Acquired Immune Deficiency Syndromes and Human Retrovirology	1
Journal of Adolescence	1
Journal of Adolescent Health	5
Journal of Adolescent Research	1
Journal of Advanced Transportation	10
Journal of Aging and Health	1
Journal of Antimicrobial Chemotherapy	2
Journal of Architecture	1
Journal of Asthma	2
Journal of Atmospheric and Solar-Terrestrial Physics	2
Journal of Bone and Mineral Research	1
Journal of Clinical Neurology	2
Journal of Clinical Psychiatry	1
Journal of Clinical Virology	3
Journal of Community Health	1
Journal of Community Psychology	4
Journal of Computational and Applied Mathematics	1
Journal of Computer and System Sciences	1
Journal of Computing in Civil Engineering	3
Journal of Construction Engineering and Management-ASCE	1
Journal of Criminal Justice	4
Journal of Cultural Economics	2
Journal of Developmental Origins of Health and Disease	1
Journal of Drug Issues	2
Journal of Economic Behavior & Organization	2
Journal of Environmental Economics and Management	2
Journal of Environmental Economics and Policy	1
Journal of Environmental Health	1

Journal title	Number of papers
Journal of Environmental Health Science and Engineering	2
Journal of Environmental Radioactivity	4
Journal of Environmental Science and Health Part B-Pesticides Food Contaminants and Agricultural Wastes	1
Journal of Epidemiology and Community Health	12
Journal of Ethnobiology and Ethnomedicine	1
Journal of European Real Estate Research	5
Journal of Exposure Analysis and Environmental Epidemiology	4
Journal of Exposure Science and Environmental Epidemiology	10
Journal of Family Issues	2
Journal of Field Robotics	5
Journal of Forensic Sciences	1
Journal of Forest Economics	1
Journal of Healthcare Engineering	1
Journal of Infectious Diseases	2
Journal of Intelligent Transportation Systems	2
Journal of International Economics	1
Journal of International Trade & Economic Development	1
Journal of Logic and Computation	1
Journal of Maps	4
Journal of Mathematical Analysis and Applications	1
Journal of Monetary Economics	1
Journal of Network and Computer Applications	2
Journal of Neurology	2
Journal of Neurology Neurosurgery and Psychiatry	2
Journal of Neurovirology	2
Journal of Nutrition	2
Journal of Occupational and Environmental Medicine	2
Journal of Occupational Medicine and Toxicology	2
Journal of Peasant Studies	1
Journal of Pediatrics	2
Journal of Photochemistry and Photobiology A-Chemistry	2
Journal of Physical Activity & Health	1
Journal of Physical Chemistry A	1
Journal of Power Sources	1
Journal of Public Administration Research and Theory	1
Journal of Public Health Management and Practice	2
Journal of Quantitative Spectroscopy & Radiative Transfer	4
Journal of Radioanalytical and Nuclear Chemistry	3

Journal title	Number of papers
Journal of Research in Crime and Delinquency	3
Journal of Rural Health	1
Journal of Social Distress and the Homeless	1
Journal of the American Dietetic Association	2
Journal of the Economic and Social History Of The Orient	1
Journal of the Franklin Institute-Engineering and Applied Mathematics	1
Journal of the International Aids Society	3
Journal of the National Cancer Institute	1
Journal of the Operational Research Society	2
Journal of Toxicology and Environmental Health-Part A-Current Issues	2
Journal of Toxicology-Clinical Toxicology	1
Journal of Traffic and Transportation Engineering-English Edition	1
Journal of Transport & Health	6
Journal of Transportation Engineering-ASCE	2
Journal of Transportation Safety & Security	1
Journal of Universal Computer Science	1
Journal of Urban Economics	4
Journal of Urban Health-Bulletin of the New York Academy of Medicine	9
Journal of Virology	1
Journal of Youth and Adolescence	5
Journals of Gerontology	1
Lancet	1
Land Economics	4
Lecture Notes in Computer Science	3
Machine Vision and Applications	1
Malaria Journal	10
Maternal and Child Health Journal	2
Mathematical and Computer Modelling	1
Mathematical Problems in Engineering	5
Medical and Veterinary Entomology	2
Medicine	5
Military Medicine	1
Mobile Networks & Applications	3
Monthly Weather Review	3
Neuroepidemiology	3
New England Journal of Medicine	1
New Journal of Physics	2
Noise & Health	2

Journal title	Number of papers
North American Journal of Economics and Finance	1
Occupational and Environmental Medicine	2
Parasitology	1
Particuology	2
Pathogens and Global Health	2
Pattern Recognition Letters	1
PeerJ Computer Science	1
Perspectives in Public Health	1
Photodermatology Photoimmunology & Photomedicine	1
Physica A-Statistical Mechanics and its Applications	2
Physical Chemistry Chemical Physics	2
Physics and Chemistry of the Earth	2
Physics in Medicine and Biology	1
PLoS Computational Biology	2
PLoS Medicine	3
PLoS Neglected Tropical Diseases	20
Political Geography	4
Population Health Management	1
Population Space and Place	2
Preventing Chronic Disease	3
Preventive Medicine	6
Preventive Veterinary Medicine	1
Proceedings of the IEEE	2
Proceedings of the Institution of Civil Engineers-Engineering Sustainability	2
Psychological Science	1
Public Administration and Development	3
Public Health Nursing	1
Public Health Nutrition	5
Pure and Applied Geophysics	2
Radiocarbon	1
Regional Science and Urban Economics	3
Research in Nursing & Health	2
Research in Transportation Economics	3
Resource and Energy Economics	2
Review of Economic Studies	1
Risk Analysis	2
Rural and Remote Health	1
Scandinavian Housing & Planning Research	1

Journal title	Number of papers
Scandinavian Journal of Public Health	1
Schizophrenia Bulletin	4
School Psychology International	2
Sexual Health	1
Simulation-Transactions of the Society for Modeling and Simulation International	2
Sleep	1
Social & Cultural Geography	3
Social Compass	1
Social History of Medicine	1
Social Indicators Research	4
Social Science & Medicine	10
Social Science Quarterly	3
Social Work in Health Care	1
Sociobiology	3
Sociological Perspectives	2
Soft Computing	3
Solid Earth	2
Sports Medicine	2
Statistics in Medicine	1
Structural Engineering International	2
Substance Use & Misuse	2
Systems Research and Behavioral Science	1
Technological Forecasting and Social Change	10
Tourism Economics	1
Tourism Management	2
Traffic Injury Prevention	3
Transactions of the Royal Society of Tropical Medicine and Hygiene	2
Transport	3
Transport Policy	4
Transportation	5
Transportation in Developing Economies	3
Transportation Planning and Technology	2
Transportation Research Part A-Policy and Practice	14
Transportation Research Part B-Methodological	7
Transportation Research Part C-Emerging Technologies	12
Transportation Research Part D-Transport and Environment	16
Transportation Research Part F-Traffic Psychology and Behaviour	4
Transportation Research Record	24

Journal title	Number of papers
Transportmetrica A-Transport Science	3
Travel Behaviour and Society	2
Tropical Medicine & International Health	2
Tunnelling and Underground Space Technology	3
Turkish Online Journal of Design Art and Communication	2
Urban Design International	2
Utilities Policy	4
Vector-Borne and Zoonotic Diseases	3
Vehicular Communications	5
Veterinary Parasitology	2
Visual Computer	1
Wireless Communications & Mobile Computing	3
Wireless Personal Communications	8
Womens Studies International Forum	2
World Development	5
Zoonoses And Public Health	2

TABLE S2. Summary of urban homogenization (UH) findings for all sources included in systematic analysis. Types of UH: taxonomic = increased species composition similarity, functional = increased functional similarity, phylogenetic = increased relatedness of communities, structural = increased similarity in physical structure, temporal = decreased temporal variability through time, historical = increased similarity over time, abiotic = increased similarity of abiotic characteristics.

Citation	Type	Sample location	Spatial scale	Taxa / Condition	UH conclusions
Aronson et al. 2014	Taxonomic	Entire cities	Multiple continents (6)	Multiple (birds & plants)	No, more clustering geographically than by urban
Aronson et al. 2015	Taxonomic	Grid cells, urban land cover gradient	1 country (USA)	Woody plants	No, beta diversity increased with urbanization for all nativity groups
Asmus and Rapson 2014	Taxonomic	Entire cities	1 country (New Zealand)	Spontaneous plants	Yes, no correlation between geographic distance and floristic similarity
Bertocci et al. 2017	Taxonomic	Natural patches in urban / non-urban	1 country (Portugal)	Multiple (macro-algae & invertebrates)	No, inconsistent evidence among time periods and locations, no overall trend of homogenization
Blair 2004*	Taxonomic	Points along urban land use gradient (6 land use types)	1 country (USA)	Birds	Yes, more urban sites have more similar communities
Blair and Johnson 2008	Taxonomic	Points along urban land use gradient (6 land use types)	1 country (USA)	Birds	Yes, more urban sites have more similar communities
Brice et al. 2017	Functional & taxonomic	Natural patches, surrounding urban land cover gradient	1 city (Canada)	Herbaceous plants	No, found greater functional and taxonomic differentiation among more urban forests
Catterall et al. 2010	Historical (taxonomic)	Points in suburban / natural	1 city (Australia)	Birds	Yes, suburbs were more similar to each other than forests, and both become more homogeneous over time but suburbs more so
Deák et al. 2016	Taxonomic	Parks / vacant lots in city center and periphery	1 city (Hungary)	Spontaneous plants	Mixed, more cosmopolitan species but not more alien species in more urban sites

Citation	Type	Sample location	Spatial scale	Taxa / Condition	UH conclusions
Docherty et al. 2018	Taxonomic	Points in urban / rural	1 country (USA)	Bacteria	Mixed, soil bacteria were more homogeneous in urban but no difference in air bacteria
Epp Schmidt et al. 2017	Taxonomic	Points along urban land use gradient (4 land use types)	Multiple continents (3)	Soil microbes	Mixed, convergence of archaea and fungi but not bacteria
Filloy et al. 2015	Taxonomic	Points in urban / rural	1 country (Argentina)	Birds	Yes, urban assemblages were more similar than rural ones, although the distance decay relationship was the same in both urban and rural
Fischer et al. 2016	Taxonomic	Parks in downtown / transition / suburb	1 city (Chile)	Spontaneous non-woody plants	No, no difference in similarity along urban to rural gradient
Garcillán et al. 2014	Taxonomic	Entire cities	Multiple continents (2)	Spontaneous plants	Yes, non-native species increase homogeneity of urban floras
Hall et al. 2016	Abiotic	Points in residential / natural	1 country (USA)	Microclimate	Yes, temperature and humidity were more similar among residential yards than among areas of native vegetation
Harrison et al. 2018	Phylogenetic & taxonomic	Points in urban / agricultural / natural	1 country (USA)	Bees	Mixed, increased phylogenetic similarity in urban relative to agriculture and forest but no difference in taxonomic similarity or distance decay
Hartley et al. 2007	Taxonomic	Parks in urban / suburban / rural	1 city (Canada)	Beetles	No, found equal heterogeneity in urban, suburban, and rural communities
Holway and Suarez 2006	Functional & taxonomic	Natural patches in urban / non-urban	1 country (USA)	Ants	Yes, greater similarity among invaded urban fragments compared to uninvaded scrub

Citation	Type	Sample location	Spatial scale	Taxa / Condition	UH conclusions
Horsák et al. 2013	Taxonomic	Points along urban land use gradient (7 land use types)	1 continent, multiple countries (Europe)	Snails	Mixed, homogenization by urban non-native species overall, but differentiation in some site types
Horsák et al. 2016*	Taxonomic	Points along urban land use gradient (7 land use types)	1 continent, multiple countries (Europe)	Snails	Mixed, native species were more similar across cities than non-native species
Huwer and Wittig 2013	Historical (taxonomic)	Entire cities	1 country (Germany)	Spontaneous plants	Mixed, homogenization for entire flora and all groups except neophytes, which differentiated over time
Jenerette et al. 2006	Abiotic	Patches in urban / agricultural / natural	1 city (USA)	Soils	Mixed, increased spatial homogeneity in urban soils at the patch scale but increased heterogeneity among urban sites
Jim and Chen 2008	Taxonomic	Urban parks, riverside parks, and street verges	1 city (Taiwan)	Trees	Mixed, homogenization of trees among riverside parks and street verges, but not urban parks
Johnson et al. 2013	Taxonomic	Natural patches in urban / agricultural / natural	1 country (USA)	Multiple (amphibians, reptiles, crayfish, mollusks, & aquatic insects)	No, no difference in community similarity among land uses
Kale et al. 2018	Temporal (taxonomic)	Points along urban land use gradient (5 land use types)	1 city (India)	Birds	Yes, low turnover between communities in the same urbanization stage
Knapp and Wittig 2012*	Historical (taxonomic)	Entire cities	1 country (Germany)	Spontaneous plants	Mixed, homogenization for entire flora and all groups except neophytes, which differentiated over time

Citation	Type	Sample location	Spatial scale	Taxa / Condition	UH conclusions
Knop 2016	Taxonomic	Points in urban / agricultural	1 country (Switzerland)	Insects (true bugs, beetles, and leafhoppers)	Mixed, homogenization of true bugs and leafhoppers but not beetles, and generalist beetles led to differentiation in cities
La Sorte and McKinney 2006*	Taxonomic	Entire cities	1 country (USA)	Plants	Mixed, native species in cities were most similar at short geographic distances but extra-limital native species had a smaller slope over distance
La Sorte et al. 2007	Taxonomic	Entire cities	Multiple continents (2)	Plants	Mixed, found higher similarity among U.S. floras and among European floras than between U.S. and European floras showing regional but not cross-continental homogenization
La Sorte et al. 2008*	Taxonomic	Entire cities	1 continent, multiple countries (Europe)	Spontaneous plants	Mixed, archeophytes were most similar with least decay over distance and neophytes were most dissimilar with most decay over distance, natives were intermediate
La Sorte et al. 2014a*	Taxonomic	Entire cities	Multiple continents (6)	Plants	Yes, urban floras had greater compositional similarity due to greater similarity of non-native compared to native species
La Sorte et al. 2014b	Temporal (taxonomic)	Grid cells, urban / agricultural / natural land cover	1 continent, multiple countries (North America)	Birds	Yes, less seasonal variation in bird composition in urban and agricultural than in natural areas
Leveau and Leveau 2016	Temporal (taxonomic)	Parks at various distances to urban center	1 city (Argentina)	Birds	Yes, less seasonal variation in bird composition in urban parks compared to parks far from the urban center

Citation	Type	Sample location	Spatial scale	Taxa / Condition	UH conclusions
Leveau et al. 2015	Temporal (taxonomic)	Points along urban land use gradient (5 land use types)	1 city (Argentina)	Birds	Yes, less seasonal variation in species richness, abundance, and composition in more urban compared to more agricultural areas
Lososová et al. 2012	Taxonomic	Points along urban land use gradient (7 land use types)	1 continent, multiple countries (Europe)	Spontaneous plants	Mixed, found differentiation by alien species overall but homogenization by archeophytes and differentiation by neophytes
Lososová et al. 2016*	Taxonomic	Points along urban land use gradient (7 land use types)	1 continent, multiple countries (Europe)	Spontaneous plants	Mixed, found that neophytes differentiated while archeophytes homogenized
Luck and Smallbone 2011	Functional & taxonomic	Points along urban development gradient	1 country (Australia)	Birds	Yes, slope of taxonomic and functional similarity over distance is smaller for urban bird communities compared to non-urban bird communities
Magura et al. 2010	Taxonomic	Natural patches in urban / suburban / rural	Multiple continents (3)	Beetles	No, urban faunas grouped with their rural counterparts and were not more similar across regions than rural faunas
Marchetti et al. 2006	Historical (taxonomic)	Watersheds, urban land cover gradient	1 country (USA)	Freshwater fish	No, differentiation of watersheds over time associated with urbanization
Morelli et al. 2016	Phylogenetic & taxonomic	Points in urban / rural	1 continent, multiple countries (Europe)	Birds	Yes, species composition was more similar among urban than rural sites and urban communities had lower evolutionary distinctiveness
Murthy et al. 2016	Taxonomic	Points in urban / non-urban	1 continent, multiple countries (North America)	Birds	Yes, slower decay in similarity with distance for urban compared to non-urban sites

Citation	Type	Sample location	Spatial scale	Taxa / Condition	UH conclusions
Newbold et al. 2018	Taxonomic	Points along gradient of land use intensity	Multiple continents (6)	Multiple (plants, invertebrates, birds, amphibians, reptiles, & mammals)	Yes, found human-dominated land uses had more species with broad distributions
Nock et al. 2013	Taxonomic	Points in urban / non-urban	1 country (USA)	Trees	Yes, urban tree communities were more similar to each other than were extra urban forest communities
Öckinger et al. 2009	Taxonomic	Points in urban / agricultural	1 city (Sweden)	Butterflies	Yes, slightly lower beta diversity in urban compared to agricultural
Oliveira et al. 2014	Taxonomic	Natural patches in urban / non-urban	1 country (Portugal)	Multiple (intertidal benthic organisms)	No, greater variability among urban communities
Padullés Cubino et al. 2019	Phylogenetic	Residential yards	1 country (USA)	Plants	Mixed, cultivated non-natives promoted phylogenetic differentiation while spontaneous non-natives promoted phylogenetic homogenization
Patitucci et al. 2011	Taxonomic	Points in urban / suburban / natural	1 country (Argentina)	Flies	No, no difference in similarity by site type
Pearse et al. 2018	Phylogenetic, taxonomic, & structural	Patches in urban / natural	1 country (USA)	Plants	Mixed, homogenization of community composition and tree structure, but phylogenetic differentiation in cultivated species
Pino et al. 2009	Taxonomic	Grid cells, land cover gradient	1 country (Spain)	Plants	No, found increasing similarity with climatic similarity and decreasing geographic distance but no homogenization with human disturbance

Citation	Type	Sample location	Spatial scale	Taxa / Condition	UH conclusions
Pool and Olden 2012	Historical (functional & taxonomic)	Watersheds, urban land cover gradient	1 country (USA)	Freshwater fish	No, found both functional and taxonomic homogenization over time but not predicted by urban land cover
Pouyat et al. 2015	Abiotic	Patches along urban land use gradient (4 land use types)	multiple continents (3)	Soils	Mixed, convergence for pH, organic carbon, and total nitrogen, but divergence for phosphorus and potassium
Reichert et al. 2017	Taxonomic	Natural patches at varying distance to urban land use	1 country (USA)	Mammals	Yes, increasing similarity closer to urban
Ricotta et al. 2014	Taxonomic	Entire cities	1 continent, multiple countries (Europe)	Spontaneous plants	Mixed, archeophytes and invasive neophytes promote homogenization while neophytes overall promote differentiation
Rocha et al. 2016	Taxonomic	Points in urban / rural / natural	1 city (Argentina)	Tardigrades	Yes, urban communities had lower beta diversity than rural or natural communities
Rogers et al. 2009	Historical (taxonomic)	Natural patches, surrounding urban land cover gradient	1 country (USA)	Understory plants	Yes, increased similarity over time of forest patches with more surrounding urban cover
Scott 2006	Taxonomic	Watersheds, development gradient	1 country (USA)	Freshwater fish	Yes, relative abundance of endemic to cosmopolitan species decreased with increasing development in watershed
Steele et al. 2014	Abiotic	Entire cities	1 country (USA)	Water	Yes, greater similarity in water body distribution among cities than among natural areas

Citation	Type	Sample location	Spatial scale	Taxa / Condition	UH conclusions
Trentanovi et al. 2013	Taxonomic	Natural patches, surrounding urban land cover gradient	1 city (Germany)	Understory plants	Mixed, homogenization of total and native species in intermediate urbanization but not high or low urbanization
Uchida et al. 2018	Temporal (taxonomic)	Natural patches, surrounding urban land cover gradient	1 city (Japan)	Multiple (plants & butterflies)	Yes, temporal beta diversity of both butterflies and plants decreased with increasing urbanization
Ulrich et al. 2007	Taxonomic	Points along urbanization gradient from city center	1 city (Poland)	Beetles	No, no evidence of homogenization for beetles in two trophic guilds
Villegas Vallejos et al. 2016	Taxonomic	Points in urban / rural / natural	1 country (Brazil)	Birds	Yes, lower beta diversity in urban compared to rural and natural sites
Vázquez-Reyes et al. 2017	Taxonomic	Points in primary forest / secondary forest / settlements	1 country (Mexico)	Birds	Yes, lower dissimilarity among human settlements than among forests
Wang et al. 2014	Taxonomic	City parks, residential areas, city squares, and roadsides	1 country (China)	Plants	Yes, homogenization by non-native species for both cultivated and wild species
Yang et al. 2015	Taxonomic	Entire cities	Multiple continents (6)	Trees	Mixed, different results by spatial scale but not global scale homogenization

Notes: Entries marked with * were not included in the final sample due to data overlap with another included paper.

TABLE S3. Urban homogenization (UH) results by taxa and subgroups of taxa. Values in the evidence columns show the number and percentage of studies with each result. Total N = 57.

Taxa	Subgroup	Evidence for UH	Mixed evidence	No evidence / evidence against UH
Amphibians - 2 studies (4%)		1 (50%) (35)	-	1 (50%) (22)
Aquatic invertebrates - 1 study (2%)		-	-	1 (100%) (22)
Birds - 14 studies (25%)		13 (93%) (5, 7, 11, 23, 26, 27, 28, 30, 33, 34, 35, 54, 55)	-	1 (7%) (1)
Fish - 3 studies (5%)		1 (33%) (49)	-	2 (66%) (32, 43)
Mammals - 2 studies (4%)		2 (100%) (35, 45)	-	-
Microbes - 2 studies (4%)		-	2 (100%) (9, 10)	-
Plants - 22 studies (39%)				
	All flora	2 (50%) (35, 52)	1 (25%) (25)	1 (25%) (42)
	All woody	-	-	1 (100%) (2)
	Spontaneous herbaceous / understory	1 (20%) (48)	1 (20%) (51)	3 (60%) (1, 6, 12)
	All spontaneous	3 (38%) (3, 13, 41)	4 (50%) (8, 19, 29, 46)	1 (13%) (56)
	Cultivated	2 (66%) (41, 56)	-	1 (33%) (39)
	Trees	1 (33%) (36)	2 (66%) (21, 57)	-
Reptiles - 2 studies (4%)		1 (50%) (35)	-	1 (50%) (22)
Rocky intertidal communities - 2 studies (4%)		-	-	2 (100%) (4, 38)
Terrestrial invertebrates -				

Taxa	Subgroup	Evidence for UH	Mixed evidence	No evidence / evidence against UH
11 studies (19%)				
	General	-	1 (100%) (35)	-
	Ants	1 (100%) (17)	-	-
	Beetles	-	-	4 (100%) (16, 24, 31, 53)
	Blowflies	-	-	1 (100%) (40)
	Butterflies	2 (100%) (37, 52)	-	-
	Leafhoppers	1 (100%) (24)	-	-
	Snails	-	1 (100%) (18)	-
	Tardigrades	1 (100%) (47)	-	-
	True bugs	1 (100%) (24)	-	-

Notes: Small numerals indicate references. (1) Aronson et al. 2014; (2) Aronson et al. 2015; (3) Asmus and Rapson 2014; (4) Bertocci et al. 2017; (5) Blair and Johnson 2008; (6) Brice et al. 2017; (7) Catterall et al. 2010; (8) Deák et al. 2016; (9) Docherty et al. 2018; (10) Epp Schmidt et al. 2017; (11) Filloy et al. 2015; (12) Fischer et al. 2016; (13) Garcillán et al. 2014; (14) Hall et al. 2016; (15) Harrison et al. 2018; (16) Hartley et al. 2007; (17) Holway and Suarez 2006; (18) Horsák et al. 2013; (19) Huwer and Wittig 2013; (20) Jenerette et al. 2006; (21) Jim and Chen 2008; (22) Johnson et al. 2013; (23) Kale et al. 2018; (24) Knop 2016; (25) La Sorte et al. 2007; (26) La Sorte et al. 2014b; (27) Leveau and Leveau 2016; (28) Leveau et al 2015; (29) Lososová et al. 2012; (30) Luck and Smallbone 2011; (31) Magura et al. 2010; (32) Marchetti et al. 2006; (33) Morelli et al. 2016; (34) Murthy et al. 2016; (35) Newbold et al. 2018; (36) Nock et al. 2013; (37) Öckinger et al. 2009; (38) Oliveira et al. 2014; (39) Padullés Cubino et al. 2019; (40) Patitucci et al. 2011; (41) Pearse et al. 2018; (42) Pino et al. 2009; (43) Pool and Olden 2012; (44) Pouyat et al. 2015; (45) Reichert et al. 2017; (46) Ricotta et al. 2014; (47) Rocha et al. 2016; (48) Rogers et al. 2009; (49) Scott 2006; (50) Steele et al. 2014; (51) Trentanovi et al. 2013; (52) Uchida et al. 2018; (53) Ulrich et al. 2007; (54) Villegas Vallejos et al. 2016; (55) Vázquez-Reyes et al. 2017; (56) Wang et al. 2014; (57) Yang et al. 2015

APPENDIX B

CHAPTER 3 SUPPLEMENTARY TABLES AND FIGURES

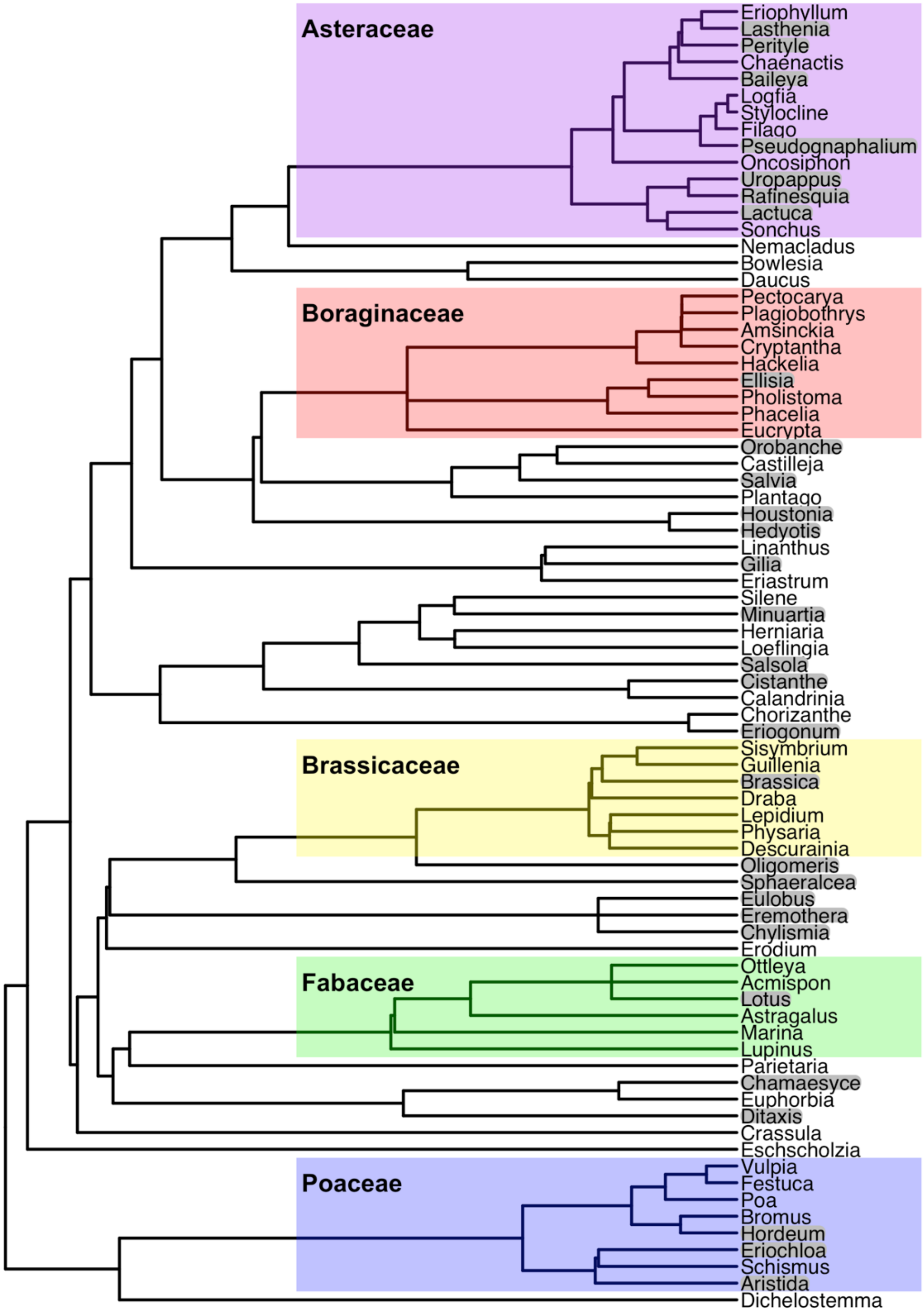

FIG. S1. Phylogenetic tree by genus for all annual plant genera observed in this study. Tree is adapted from the ALLMB tree defined by Smith and Brown (2018). Colors represent family groupings. Species found in less than 0.5% of all sampled plot (<5 plots of 952) and that were omitted from Fig. 3.02 are shown with grey backgrounds.

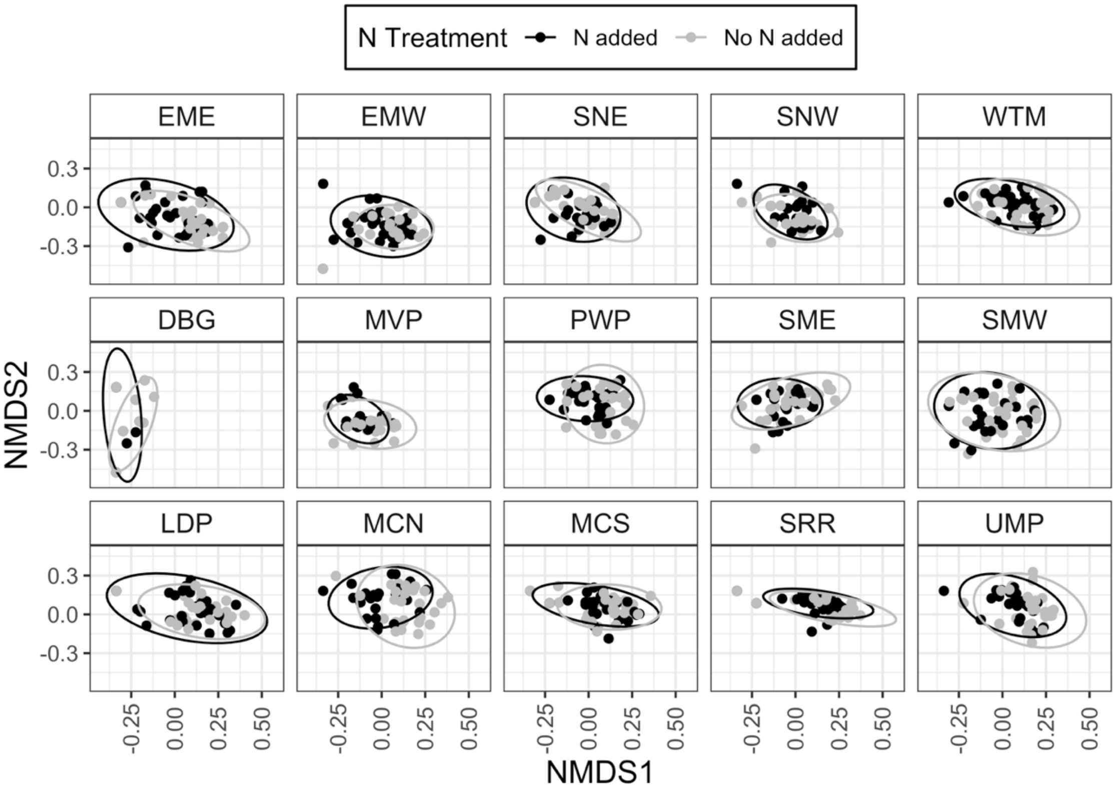

FIG. S2. NMDS of presence/absence data at the genus level for all samples, colored by N enrichment treatment and separated by site. Sites in the top row are non-urban (West), middle row are urban, and bottom row are non-urban (East). Ellipses show 95% confidence intervals. See Table 3.1 for site descriptions.

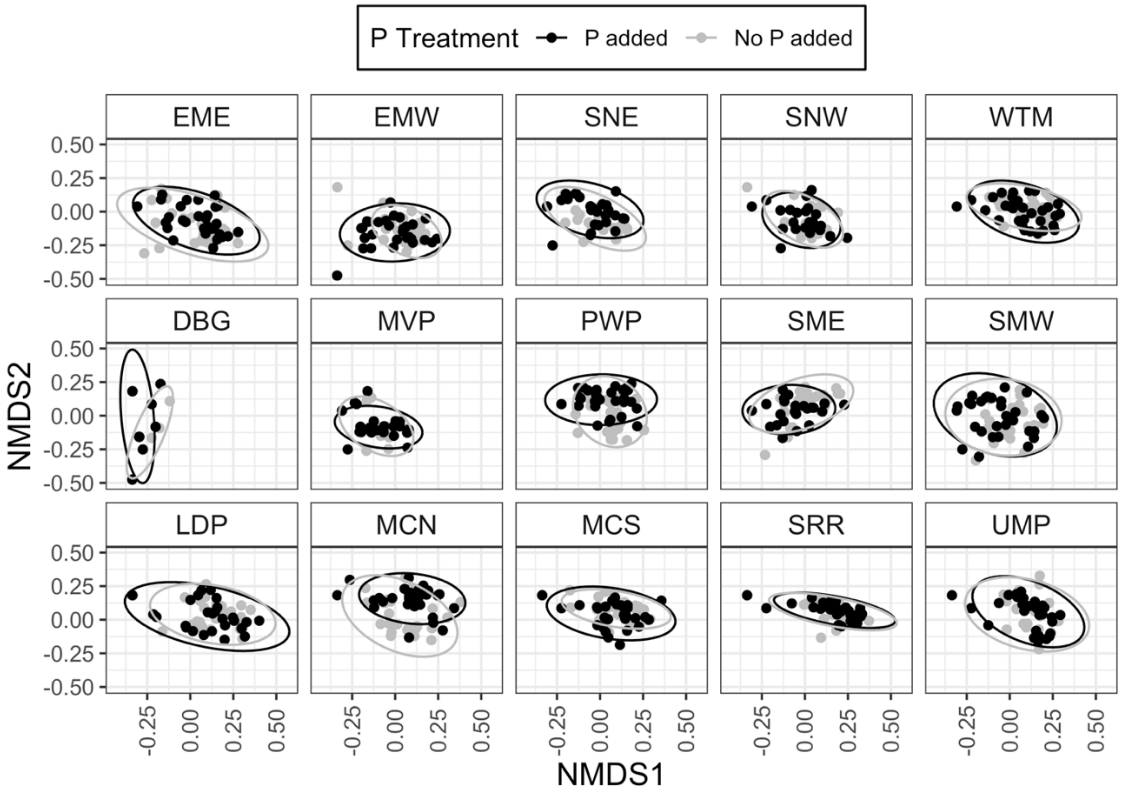

FIG. S3. NMDS of presence/absence data at the genus level for all samples, colored by P enrichment treatment and separated by site. Sites in the top row are non-urban (West), middle row are urban, and bottom row are non-urban (East). Ellipses show 95% confidence intervals. See Table 3.1 for site descriptions.

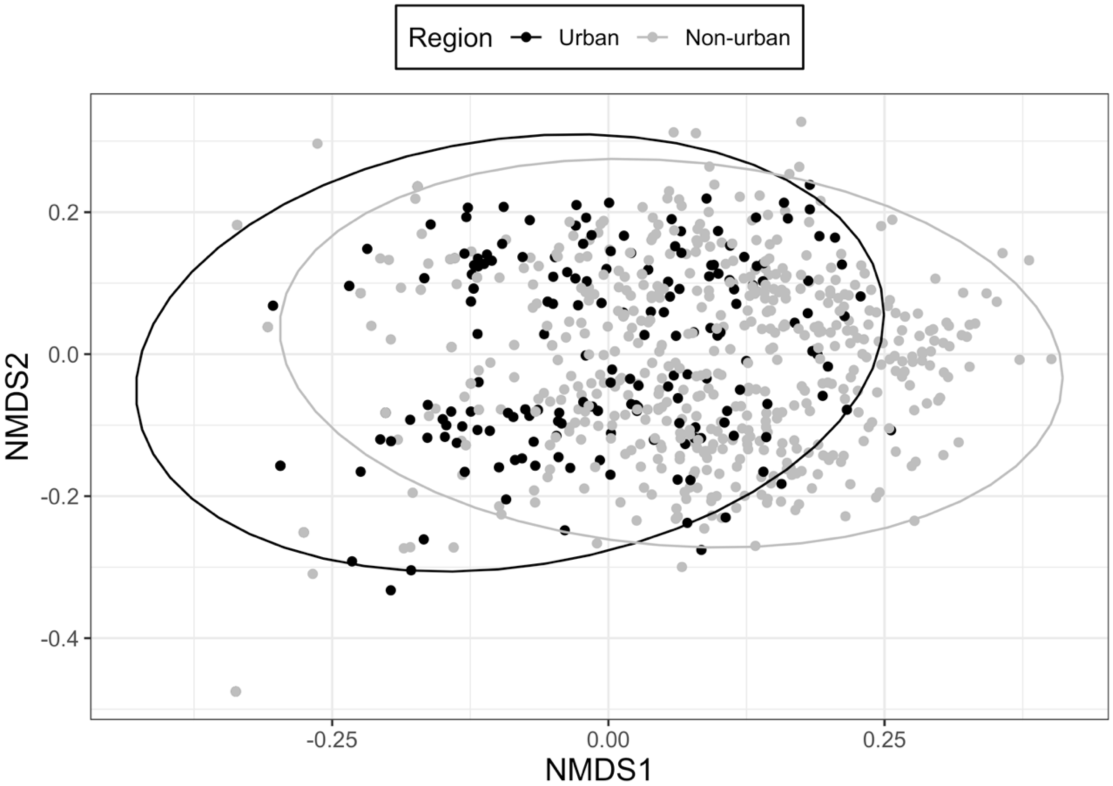

FIG. S4. NMDS of presence/absence data at the genus level for all samples, colored by urban location. Ellipses show 95% confidence intervals.

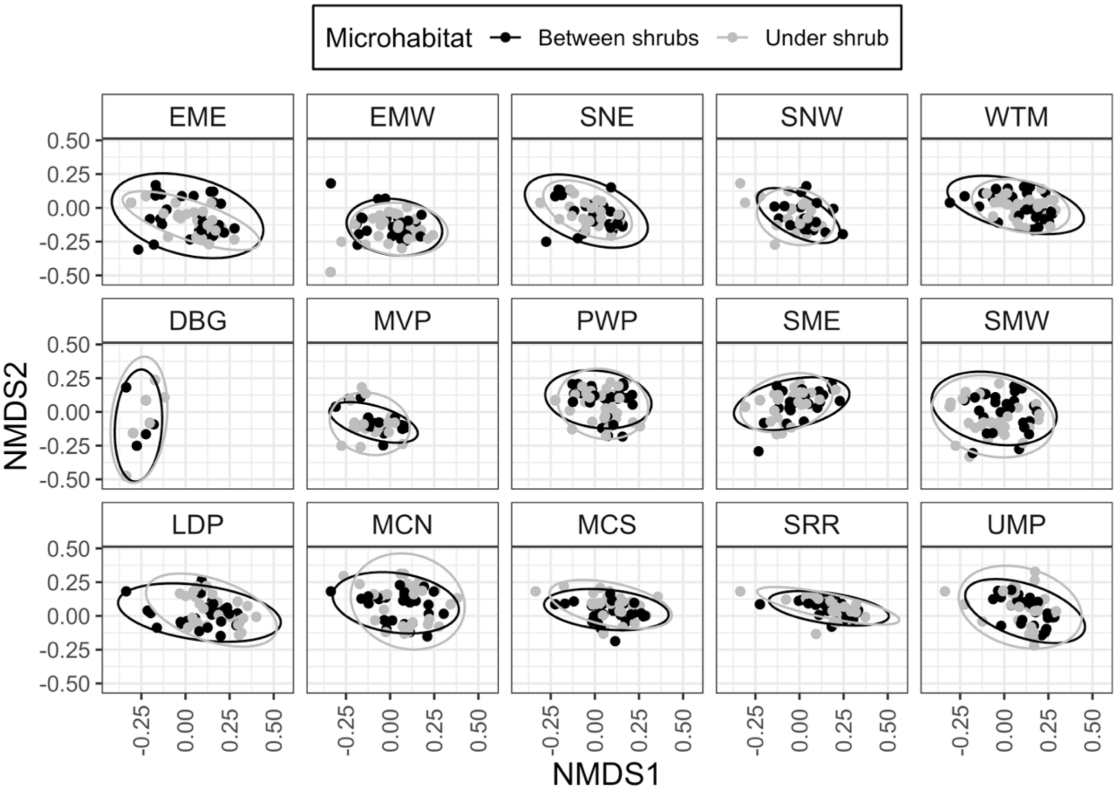

FIG. S5. NMDS of presence/absence data at the genus level for all samples, colored by microhabitat and separated by site. Sites in the top row are non-urban (West), middle row are urban, and bottom row are non-urban (East). Ellipses show 95% confidence intervals. See Table 3.1 for site descriptions.

TABLE S1. The frequency and nativity classification of all species recorded in any plot.

Family	Genus	Species[1]	Nativity[2]	Number of samples[3]
Amaranthaceae	*Salsola*	*Salsola kali*	NN	1
Apiaceae	*Bowlesia*	*Bowlesia incana*	N	62
Apiaceae	*Daucus*	*Daucus pusillus*	N	29
Asparagaceae	*Dichelostemma*	*Dichelostemma* sp.	N	1
Asparagaceae	*Dichelostemma*	*Dichelostemma capitatum*	N	5
Boraginaceae	*Amsinckia*	*Amsinckia* sp.	N	350
Boraginaceae	*Amsinckia*	*Amsinckia menziesii*	N	90
Boraginaceae	*Cryptantha*	*Cryptantha* sp.	N	32
Boraginaceae	*Cryptantha*	*Cryptantha angustifolia*	N	80
Boraginaceae	*Cryptantha*	*Cryptantha barbigera*	N	5
Boraginaceae	*Cryptantha*	*Cryptantha decipiens*	N	204
Boraginaceae	*Cryptantha*	*Cryptantha maritima*	N	66
Boraginaceae	*Cryptantha*	*Cryptantha pterocarya*	N	4
Boraginaceae	*Eucrypta*	*Eucrypta* sp.	N	58
Boraginaceae	*Eucrypta*	*Eucrypta chrysanthemifolia*	N	6
Boraginaceae	*Hackelia*	*Hackelia floribunda*	N	6
Boraginaceae	*Pectocarya*	*Pectocarya* sp.	N	859
Boraginaceae	*Phacelia*	*Phacelia* sp.	N	1
Boraginaceae	*Phacelia*	*Phacelia crenulata*	N	1
Boraginaceae	*Phacelia*	*Phacelia distans*	N	27
Boraginaceae	*Pholistoma*	*Pholistoma auritum*	N	11
Boraginaceae	*Plagiobothrys*	*Plagiobothrys arizonicus*	N	109
Boraginaceae		Unidentified		3
Brassicaceae	*Brassica*	*Brassica* sp.	NN	1
Brassicaceae	*Descurainia*	*Descurainia* sp.	M	12
Brassicaceae	*Descurainia*	*Descurainia pinnata*	N	4
Brassicaceae	*Draba*	*Draba cuneifolia*	N	99
Brassicaceae	*Guillenia*	*Guillenia lasiophylla*	N	96
Brassicaceae	*Lepidium*	*Lepidium lasiocarpum*	N	359
Brassicaceae	*Physaria*	*Physaria gordonii*	N	102
Brassicaceae	*Sisymbrium*	*Sisymbrium irio*	NN	20
Brassicaceae		Unidentified		1
Campanulaceae	*Nemacladus*	*Nemacladus glanduliferus*	N	7
Campanulaceae	*Nemacladus*	*Nemacladus rigidus*	N	4
Caryophyllaceae	*Herniaria*	*Herniaria hirsuta*	NN	31
Caryophyllaceae	*Loeflingia*	*Loeflingia squarrosa*	N	27
Caryophyllaceae	*Minuartia*	*Minuartia douglasii*	N	2
Caryophyllaceae	*Silene*	*Silene antirrhina*	N	15
Compositae	*Baileya*	*Baileya multiradiata*	N	1
Compositae	*Chaenactis*	*Chaenactis stevioides*	N	44
Compositae	*Eriophyllum*	*Eriophyllum* sp.	N	5

Family	Genus	Species[1]	Nativity[2]	Number of samples[3]
Compositae	*Eriophyllum*	*Eriophyllum lanosum*	N	123
Compositae	*Filago*	*Filago depressa*	N	7
Compositae	*Lactuca*	*Lactuca serriola*	NN	3
Compositae	*Lasthenia*	*Lasthenia californica*	N	4
Compositae	*Logfia*	*Logfia arizonica*	N	323
Compositae	*Logfia*	*Logfia filaginoides*	N	1
Compositae	*Oncosiphon*	*Oncosiphon piluliferum*	NN	23
Compositae	*Perityle*	*Perityle emoryi*	N	1
Compositae	*Pseudognaphalium*	*Pseudognaphalium*	M	1
Compositae	*Rafinesquia*	*Rafinesquia neomexicana*	N	4
Compositae	*Sonchus*	*Sonchus oleraceus*	NN	12
Compositae	*Stylocline*	*Stylocline gnaphaloides*	N	2
Compositae	*Stylocline*	*Stylocline micropoides*	N	43
Compositae	*Uropappus*	*Uropappus lindleyi*	N	2
Compositae		Unidentified		2
Crassulaceae	*Crassula*	*Crassula connata*	N	272
Euphorbiaceae	*Chamaesyce*	*Chamaesyce* sp.	N	3
Euphorbiaceae	*Ditaxis*	*Ditaxis lanceolata*	N	1
Euphorbiaceae	*Ditaxis*	*Ditaxis neomexicana*	N	1
Euphorbiaceae	*Euphorbia*	*Euphorbia albomarginata*	N	23
Euphorbiaceae	*Euphorbia*	*Euphorbia polycarpa*	N	33
Geraniaceae	*Erodium*	*Erodium cicutarium*	NN	175
Geraniaceae	*Erodium*	*Erodium texanum*	N	129
Hydrophyllaceae	*Ellisia*	*Ellisia* sect. *Eucrypta*	N	2
Lamiaceae	*Salvia*	*Salvia columbariae*	N	4
Leguminosae	*Acmispon*	*Acmispon brachycarpus*	N	44
Leguminosae	*Acmispon*	*Acmispon maritimus*	N	130
Leguminosae	*Astragalus*	*Astragalus* sp.	M	77
Leguminosae	*Astragalus*	*Astragalus nuttallianus*	N	10
Leguminosae	*Lotus*	*Lotus* sp.	M	2
Leguminosae	*Lupinus*	*Lupinus concinnus*	N	4
Leguminosae	*Lupinus*	*Lupinus sparsiflorus*	N	62
Leguminosae	*Marina*	*Marina* sp.	N	20
Leguminosae	*Ottleya*	*Ottleya strigosa*	N	102
Malvaceae	*Sphaeralcea*	*Sphaeralcea coulteri*	N	2
Onagraceae	*Chylismia*	*Chylismia claviformis*	N	2
Onagraceae	*Eremothera*	*Eremothera chamaenerioides*	N	2
Onagraceae	*Eulobus*	*Eulobus californicus*	N	1
Orobanchaceae	*Castilleja*	*Castilleja exserta*	N	12
Orobanchaceae	*Orobanche*	*Orobanche cooperi*	N	1
Papaveraceae	*Eschscholzia*	*Eschscholzia arizonica*	N	5
Papaveraceae	*Eschscholzia*	*Eschscholzia californica*	N	46

Family	Genus	Species[1]	Nativity[2]	Number of samples[3]
Plantaginaceae	*Plantago*	*Plantago* sp.	M	3
Plantaginaceae	*Plantago*	*Plantago ovata*	N	308
Plantaginaceae	*Plantago*	*Plantago patagonica*	N	163
Poaceae	*Aristida*	*Aristida adscensionis*	N	2
Poaceae	*Bromus*	*Bromus* sp.	M	1
Poaceae	*Bromus*	*Bromus rubens*	NN	79
Poaceae	*Eriochloa*	*Eriochloa aristata*	N	1
Poaceae	*Festuca*	*Festuca* sp.	M	5
Poaceae	*Hordeum*	*Hordeum murinum*	NN	1
Poaceae	*Poa*	*Poa bigelovii*	N	19
Poaceae	*Schismus*	*Schismus arabicus*	NN	856
Poaceae	*Vulpia*	*Vulpia microstachys*	N	9
Poaceae	*Vulpia*	*Vulpia octoflora*	N	260
Polemoniaceae	*Eriastrum*	*Eriastrum diffusum*	N	48
Polemoniaceae	*Eriastrum*	*Eriastrum eremicum*	N	12
Polemoniaceae	*Gilia*	*Gilia stellata*	N	1
Polemoniaceae	*Linanthus*	*Linanthus* sp.	N	1
Polemoniaceae	*Linanthus*	*Linanthus bigelovii*	N	28
Polemoniaceae	*Linanthus*	*Linanthus demissus*	N	15
Polemoniaceae		Unidentified		1
Polygonaceae	*Chorizanthe*	*Chorizanthe* sp.	N	48
Polygonaceae	*Chorizanthe*	*Chorizanthe rigida brevicornu*	N	7
Polygonaceae	*Eriogonum*	*Eriogonum thomasii*	N	1
Portulacaceae	*Calandrinia*	*Calandrinia* sp.	N	1
Portulacaceae	*Calandrinia*	*Calandrinia ciliata*	N	9
Portulacaceae	*Cistanthe*	*Cistanthe monandra*	N	1
Resedaceae	*Oligomeris*	*Oligomeris linifolia*	N	1
Rubiaceae	*Hedyotis*	*Hedyotis* sp.	N	2
Rubiaceae	*Houstonia*	*Houstonia pusilla*	N	1
Urticaceae	*Parietaria*	*Parietaria hespera*	N	9
		Unidentified		53

[1] Where only a genus is given as the species name, individuals could not be identified to species.

[2] Classifications of species nativity are for Arizona and follow the USDA Plants database. N = native, NN = non-native, and M = mixed. No nativity is assigned for taxa not identified to genus. Taxa identified to genus only were considered native unless the database showed that non-native species of the genus were also present in Arizona.

[3] Number of samples is the number of samples in which the species was observed, out of 952 total samples observed.

TABLE S2. Weather stations used to obtain precipitation and temperature data for 2006-2018.

Site ID	Data used[1]	Station manager[2]	Station name	Station ID	Site to station distance: Ground distance (km)[3]	Site to station distance: Elevation distance (m)[4]	Install date (m/d/y)[5]	Station priority[6]
DBG	P	CAP	CAP weather station	NA	0	0	10/27/06	1
DBG	P	FCDMC	Papago Park	4500	2.4	22	8/15/90	2
DBG	P	FCDMC	IBW @ McKellips Rd.	55700	2.8	33	7/15/85	2
DBG	P	FCDMC	Salt R @ Priest	67300	3.1	48	9/21/95	2
EME	P	FCDMC	Gila R@116th	70700	3.4	40	1/26/89	1
EME	P	FCDMC	Gila R. @ Estrella Pkwy.	83800	8.9	51	2/28/89	1
EME	P	FCDMC	Estrella Fan	84200	9.2	-112	11/15/92	1
EME	P	FCDMC	Agua Fria R. @ Buckeye Rd.	85000	9.6	36	10/6/88	1
EMW	P	FCDMC	Estrella Fan	84200	6.3	-62	11/15/92	1
EMW	P	FCDMC	Waterman Wash @ Rainbow Valley Rd.	83300	8	71	3/18/99	1
EMW	P	FCDMC	Gila R. @ Estrella Pkwy.	83800	9.5	101	2/28/89	1
EMW	P	FCDMC	Tuthill Rd. @ Ray Rd.	83500	10.3	92	12/22/94	1
EMW	P	FCDMC	Waterman Wash	84000	10.8	-19	5/10/83	1
LDP	P	CAP	CAP weather station	NA	0	0	5/10/06	1
LDP	P	FCDMC	Wolverine Pass	38300	4.5	-5	7/12/07	2
LDP	P	FCDMC	Apache Junction FRS	81300	6.4	68	12/16/81	2
LDP	P	FCDMC	Apache Trail	36500	6.8	89	4/14/93	2
MCN	P	FCDMC	Asher Hills	75500	1.4	33	8/2/90	1
MCN	P	FCDMC	McDowell Mountain Park	75800	5.2	-107	8/6/90	1
MCN	P	FCDMC	McDowell Mountain Road	76000	7.6	41	5/18/04	1
MCS	P	FCDMC	McDowell Mountain Road	76000	1.3	-23	5/18/04	1
MCS	P	FCDMC	Hesperus Dam	79000	4.7	-102	12/18/96	1
MCS	P	FCDMC	Fountain Hills Fire Dept.	76700	4.8	-32	12/9/93	1
MCS	P	FCDMC	Lost Dog Wash	58800	12	-24	7/13/90	1
MVP	P	FCDMC	Phoenix West Park Dam	13500	0.6	-3	11/29/01	1

					Site to station distance			
Site ID	Data used[1]	Station manager[2]	Station name	Station ID	Ground distance (km)[3]	Elevation distance (m)[4]	Install date (m/d/y)[5]	Station priority[6]
MVP	P	FCDMC	ACDC @ Cave Creek	12000	2.2	22	3/11/97	1
MVP	P	FCDMC	10th St. Wash Basin #1	11000	2.5	-5	10/23/96	1
MVP	P	FCDMC	Phoenix Basin #3	11500	3.4	-19	12/18/01	1
MVP	P	FCDMC	Dreamy Draw Dam	9800	5	-31	1/24/84	1
PWP	P	FCDMC	Phoenix Dam #99	12700	0.9	13	7/7/09	1
PWP	P	FCDMC	Dreamy Draw Dam	9800	1.7	28	1/24/84	1
PWP	P	FCDMC	Tatum basin Inflow	58000	3.2	28	6/3/94	1
PWP	P	FCDMC	Phoenix Basin #3	11500	3.8	40	12/18/01	1
SME	P	FCDMC	Guadalupe FRS	68200	1.4	-19	6/29/89	1
SME	P	FCDMC	Salt R. @ 40th St.	67000	5.9	31	3/22/96	1
SME	P	FCDMC	ASU South	67500	6.7	17	7/14/95	1
SMW	P	FCDMC	South Mountain Park HQ	69000	0.9	28	5/1/97	1
SMW	P	FCDMC	Dobbins Rd. @ 19th Ave.	68900	3.6	111	9/15/16	1
SMW	P	FCDMC	South Mountain Fan	70500	6	71	6/9/93	1
SMW	P	FCDMC	Cesar Chavez Park	66500	6.8	133	8/15/90	1
SNE	P	FCDMC	Maricopa Mountains	43000	8.5	123	4/21/05	1
SNE	P	FCDMC	Upper Waterman Wash	84500	9.3	12	6/23/88	1
SNW	P	FCDMC	Maricopa Mountains	43000	2.8	13	4/21/05	1
SNW	P	FCDMC	Bender Wash	40500	12.1	14	1/12/82	2
SNW	P	FCDMC	Upper Waterman Wash	84500	13.2	-99	6/23/88	2
SNW	P	FCDMC	Gila Bend Landfill	42500	13.4	153	4/7/93	2
SRR	P	FCDMC	Saguaro Lake	63500	4.9	-38	1/24/00	1
SRR	P	FCDMC	Bulldog Canyon	38800	8.1	-200	7/11/07	2
SRR	P	FCDMC	Usery Park WS	80700	8.2	-210	2/24/94	2
SRR	P	FCDMC	Granite Reef	75000	10.7	29	7/21/05	2
UMP	P	FCDMC	Usery Mountain Park	80200	1	19	6/20/85	1
UMP	P	FCDMC	Usery Park WS	80700	1.5	-53	2/24/94	1

					Site to station distance			
Site ID	Data used[1]	Station manager[2]	Station name	Station ID	Ground distance (km)[3]	Elevation distance (m)[4]	Install date (m/d/y)[5]	Station priority[6]
UMP	P	FCDMC	McDowell Rd. @ Hawes Rd.	33200	3.2	58	4/13/06	1
UMP	P	FCDMC	McDowell Rd. @ Meridian Rd.	33500	3.7	9	4/23/06	1
WTM	P	FCDMC	Ford Cyn Wash	86200	0.8	6	2/5/02	1
WTM	P	FCDMC	McMicken Dam South	86500	2.8	39	2/13/02	1
WTM	P	FCDMC	McMicken Dam @ Bell Rd.	71500	5.2	42	9/8/16	1
WTM	P	FCDMC	White Tank FRS #3	87300	8.2	83	3/12/86	1
DBG	T	CAP	CAP weather station	NA	0	0	10/27/06	1
DBG	T	FCDMC	Osborne at 64th St	4617	3	13	1/16/98	2
DBG	T	FCDMC	Gateway Community College	4507	5	51	12/16/03	3
EME	T	FCDMC	Estrella Fan	6892	9	-112	11/15/92	1
EME	T	NCDC	Laveen 3 SSE	USC 00024829	15	-15	7/1/48	1
EMW	T	FCDMC	Estrella Fan	6892	6	-62	11/15/92	1
EMW	T	NCDC	Litchfield Park	USC 00024977	22	65	8/1/17	2
LDP	T	CAP	CAP weather station	NA	0	0	5/10/06	1
LDP	T	NCDC	Apache Junction 5 NE	USC 00020288	1	-11	5/1/87	2
MCN	T	FCDMC	Fountain Hills Fire Station	5952	13	32	7/23/97	1
MCN	T	NCDC	Bartlett Dam	USC 00020632	12	36	9/1/39	1
MCN	T	NCDC	Fountain Hills	USC 00023190	15	58	10/1/79	2
MCS	T	FCDMC	Fountain Hills Fire Station	5952	5	-32	7/23/97	1
MCS	T	NCDC	Fountain Hills	USC 00023190	5	-6	10/1/79	1
MVP	T	NCDC	Phoenix Deer Valley Municipal Airport, AZ US	USW 00003184	12	-46	9/1/98	1
MVP	T	FCDMC	Phoenix Dam 2B	4797	4	-37	6/30/09	1
MVP	T	FCDMC	Osborne at 64th St	4617	17	15	1/16/98	2
PWP	T	NCDC	Scottsdale Municipal Airport, AZ US	USW 00003192	11	7	11/1/01	1

					Site to station distance			
Site ID	Data used[1]	Station manager[2]	Station name	Station ID	Ground distance (km)[3]	Elevation distance (m)[4]	Install date (m/d/y)[5]	Station priority[6]
PWP	T	FCDMC	Phoenix Dam 2B	4797	6	22	6/30/09	1
PWP	T	FCDMC	Osborne at 64th St	4617	11	74	1/16/98	2
SME	T	FCDMC	Gateway Community College	4507	9	23	12/16/03	1
SME	T	NCDC	Phoenix Airport, AZ US	USW 00023183	6	35	6/1/33	1
SMW	T	FCDMC	South Mountain Fan	6562	6	71	6/9/93	1
SMW	T	NCDC	Laveen 3 SSE	USC 00024829	7	112	7/1/48	2
SNE	T	FCDMC	Mobile	6972	18	83	4/3/08	1
SNE	T	NCDC	Gila Bend 3 ENE, AZ US	USW 00053176	22	254	9/25/10	2
SNE	T	FCDMC	Gila Bend Landfill	6912	22	263	1/17/02	2
SNE	T	NCDC	Gila Bend, AZ US	USC 00023393	27	268	12/1/92	2
SNW	T	FCDMC	Mobile	6972	27	-27	12/15/04	1
SNW	T	NCDC	Gila Bend 3 ENE, AZ US	USW 00053176	18	144	9/25/10	2
SNW	T	FCDMC	Gila Bend Landfill	6912	14	153	1/17/02	2
SNW	T	NCDC	Gila Bend, AZ US	USC 00023393	14	158	12/1/92	2
SRR	T	NCDC	Fountain Hills	USC 00023190	14	-47	10/1/79	1
SRR	T	NCDC	Stewart Mountain Dam, AZ US	USC 00028214	5	1	6/1/39	1
SRR	T	FCDMC	Usery Park	6652	8	-210	2/24/94	2
UMP	T	FCDMC	Usery Park	6652	1.5	-53	2/24/94	1
UMP	T	NCDC	East Mesa, AZ US	USC 00022782	6	129	8/1/02	2
WTM	T	FCDMC	McMicken Dam	5437	11	42	3/24/83	1
WTM	T	NCDC	Youngtown, AZ US	USC 00029634	18	108	10/1/64	2
WTM	T	FCDMC	Camelback Rd. @ Citrus Rd.	5417	12	114	1/22/15	2

[1] Use of either precipitation (P) or temperature (T) data from a weather station.
[2] Weather station managers include the Central Arizona-Phoenix Long Term Ecological Research project (CAP), Flood Control District of Maricopa County (FCDMP), and National Climate Data Center (NCDC).
[3] The approximate distance from each experimental site to each matched climate station
[4] The difference in elevation from each experimental site to each matched climate station.
[5] Data are available for each weather station after the listed install date, with occasional exceptions.
[6] The order in which each station was used for each site. Level 1 stations were averaged when available, with level 2 stations averaged when no level 1 stations were available for a given day, and level 3 stations (DBG temperature only) averaged when no level 1 or 2 stations were available.

TABLE S3. Number of days in which each set of weather stations from Appendix B: Table S2 was used for the growing seasons of 2007-2018 (e.g., October 2006 - March 2007 for 2007 growing season; total of 2187 days).

				Ground distance (km)[4]		Elevation distance (m)[5]		
Site ID	Type[1]	# Stations[2]	Source priority[3]	To closest station	To farthest station	To closest station	To farthest station	# Days
DBG	P	1	1	0	0	0	0	1981
DBG	P	3	2	2.4	3.1	22	48	206
EME	P	4	1	3.4	9.6	-112	51	2158
EME	P	3	1	3.4	9.6	36	51	20
EME	P	3	1	3.4	9.6	-112	40	5
EME	P	3	1	3.4	9.2	-112	51	4
EMW	P	5	1	6.3	10.8	-62	101	2051
EMW	P	4	1	6.3	10.3	-62	101	104
EMW	P	4	1	8	10.8	-19	101	13
EMW	P	3	1	8	10.3	71	101	7
EMW	P	4	1	6.3	10.8	-62	101	7
EMW	P	4	1	6.3	10.8	-62	92	5
LDP	P	1	1	0	0	0	0	2084
LDP	P	3	2	4.5	6.8	-5	89	103
MCN	P	3	1	1.4	7.6	-107	41	2175
MCN	P	2	1	1.4	5.2	-107	33	8
MCN	P	1	1	7.6	7.6	41	41	4
MCS	P	4	1	1.3	12	-102	-23	2166
MCS	P	3	1	1.3	12	-32	-23	8
MCS	P	3	1	4.7	12	-102	-24	8
MCS	P	3	1	1.3	12	-102	-23	4
MCS	P	3	1	1.3	4.8	-102	-23	1
MVP	P	5	1	0.6	5	-31	22	2177
MVP	P	4	1	0.6	5	-31	-3	4
MVP	P	4	1	2.2	5	-31	22	3
MVP	P	4	1	0.6	3.4	-19	22	2
MVP	P	4	1	0.6	5	-31	22	1
PWP	P	4	1	0.9	3.8	13	40	1632
PWP	P	3	1	1.7	3.8	28	40	544
PWP	P	3	1	0.9	3.8	13	40	7
PWP	P	2	1	1.7	3.8	28	40	3
PWP	P	3	1	0.9	3.2	13	28	1
SME	P	3	1	1.4	6.7	-19	31	2164
SME	P	2	1	1.4	5.9	-19	31	13

				Ground distance (km)[4]		Elevation distance (m)[5]		
Site ID	Type[1]	# Stations[2]	Source priority[3]	To closest station	To farthest station	To closest station	To farthest station	# Days
SME	P	2	1	1.4	6.7	-19	17	6
SME	P	2	1	5.9	6.7	17	31	4
SMW	P	3	1	0.9	6.8	28	133	1807
SMW	P	4	1	0.9	6.8	28	133	364
SMW	P	2	1	0.9	6.8	28	133	10
SMW	P	2	1	6	6.8	71	133	3
SMW	P	2	1	0.9	6	28	71	2
SMW	P	1	1	6.8	6.8	133	133	1
SNE	P	2	1	8.5	9.3	12	123	2153
SNE	P	1	1	9.3	9.3	12	12	28
SNE	P	1	1	8.5	8.5	123	123	6
SNW	P	1	1	2.8	2.8	13	13	2159
SNW	P	3	2	12.1	13.4	-99	153	20
SNW	P	2	2	13.2	13.4	-99	153	8
SRR	P	1	1	4.9	4.9	-38	-38	2177
SRR	P	3	2	8.1	10.7	-210	29	10
UMP	P	4	1	1	3.7	-53	58	2167
UMP	P	3	1	1.5	3.7	-53	58	9
UMP	P	3	1	1	3.2	-53	58	8
UMP	P	3	1	1	3.7	9	58	2
UMP	P	3	1	1	3.7	-53	19	1
WTM	P	3	1	0.8	8.2	6	83	1808
WTM	P	4	1	0.8	8.2	6	83	364
WTM	P	2	1	0.8	8.2	6	83	6
WTM	P	2	1	0.8	2.8	6	39	5
WTM	P	2	1	2.8	8.2	39	83	4
DBG	T	1	1	0	0	0	0	1795
DBG	T	1	2	3	3	13	13	391
DBG	T	1	3	5	5	51	51	1
EME	T	2	1	9	15	-112	-15	2048
EME	T	1	1	9	9	-112	-112	113
EME	T	1	1	15	15	-15	-15	19
EMW	T	1	1	6	6	-62	-62	2161
EMW	T	1	2	22	22	65	65	23
LDP	T	1	1	0	0	0	0	2186
LDP	T	1	2	1	1	-11	-11	1
MCN	T	1	1	13	13	32	32	1410

Site ID	Type[1]	# Stations[2]	Source priority[3]	Ground distance (km)[4] To closest station	Ground distance (km)[4] To farthest station	Elevation distance (m)[5] To closest station	Elevation distance (m)[5] To farthest station	# Days
MCN	T	2	1	12	13	32	36	766
MCN	T	1	2	15	15	58	58	4
MCN	T	1	1	12	12	36	36	4
MCS	T	2	1	5	5	-32	-6	1874
MCS	T	1	1	5	5	-32	-32	302
MCS	T	1	1	5	5	-6	-6	8
MVP	T	2	1	4	12	-46	-37	1629
MVP	T	1	1	12	12	-46	-46	551
MVP	T	1	1	4	4	-37	-37	4
MVP	T	1	2	17	17	15	15	3
PWP	T	2	1	6	11	7	22	1630
PWP	T	1	1	11	11	7	7	554
PWP	T	1	1	6	6	22	22	3
SME	T	2	1	6	9	23	35	2103
SME	T	1	1	6	6	35	35	84
SMW	T	1	1	6	6	71	71	2175
SMW	T	1	2	7	7	112	112	7
SNE	T	1	1	18	18	83	83	1810
SNE	T	2	2	22	27	263	268	365
SNE	T	1	2	22	22	263	263	7
SNW	T	1	1	27	27	-27	-27	1810
SNW	T	2	2	14	14	153	158	365
SNW	T	1	2	14	14	153	153	7
SRR	T	2	1	5	14	-47	1	1103
SRR	T	1	1	14	14	-47	-47	779
SRR	T	1	2	8	8	-210	-210	157
SRR	T	1	1	5	5	1	1	148
UMP	T	1	1	1.5	1.5	-53	-53	2182
UMP	T	1	2	6	6	129	129	5
WTM	T	1	1	11	11	42	42	2171
WTM	T	2	2	12	18	108	114	10
WTM	T	1	2	18	18	108	108	3

[1] Precipitation (P) or temperature (T) data.
[2] Number of stations for which daily data were averaged.
[3] The station priority level of the available data (see Appendix B: Table S2).
[4] The approximate distance from each experimental site to each matched climate station
[5] The difference in elevation from each experimental site to each matched climate station.

TABLE S4. Eight species richness component models with Akaike weights > 0.01.

Model	df	logLik	AICc	Delta AIC	Model weight
water + microhabitat + N + previous-water + urban + water:previous-water + water:urban + previous-water:urban	10	-2314.82	4649.88	0.00	0.22
water + microhabitat + N + P + previous-water + urban + water:previous-water + water:urban + P:urban + previous-water:urban	12	-2313.20	4650.73	0.84	0.14
water + microhabitat + N + P + previous-water + urban + water:previous-water + water:urban + previous-water:urban	11	-2314.28	4650.83	0.95	0.14
water + microhabitat + N + previous-water + urban + water: microhabitat + water:previous-water + water:urban + previous-water:urban	11	-2314.40	4651.09	1.20	0.12
water + microhabitat + N + previous-water + urban + water:previous-water + water:urban + microhabitat:urban + previous-water:urban	11	-2314.48	4651.25	1.37	0.11
water + microhabitat + N + previous-water + urban + water:N + water:previous-water + water:urban + previous-water:urban	11	-2314.64	4651.56	1.68	0.09
water + microhabitat + N + previous-water + urban + water:previous-water + water:urban + N:urban + previous-water:urban	11	-2314.64	4651.57	1.69	0.09
water + microhabitat + N + P + previous-water + urban + water:P + water:previous-water + water:urban + previous-water:urban	12	-2313.70	4651.73	1.85	0.09

Notes: Models predict species richness with site as a random factor and all two-way interaction terms included in the global model. Terms included in each submodel are water = current growing season water availability, microhabitat = microhabitat under or between shrubs, N = nitrogen addition, P = phosphorus addition, previous-water = previous growing season water availability, and urban = site location within or outside the urban core.

TABLE S5. SEM regression coefficients.

Response	Predictor	Estimate	Std error	Std estimate	p value
Taxonomic diversity[1]					
$R^2 = 0.309$	N	-2.0	0.2	-0.28	<0.001
	P	-0.3	0.2	-0.04	0.22
	Microhabitat (shrub)	0.8	0.2	0.11	0.001
	Urban	-2.3	0.3	-0.31	<0.001
	Current growing season water availability	8	1	0.31	<0.001
	Previous growing season water availability	-4	1	-0.10	0.007
	ANPP	0.02	0.01	0.06	0.14
Phylogenetic diversity[2]					
$R^2 = 0.106$	N	-0.01	0.08	0.00	0.92
	P	0.07	0.08	0.03	0.37
	Microhabitat (shrub)	-0.65	0.08	-0.31	<0.001
	Urban	0.15	0.08	0.07	0.07
	Current growing season water availability	0.5	0.3	0.06	0.17
	Previous growing season water availability	0.6	0.4	0.06	0.15
ANPP					
$R^2 = 0.372$	N	6.5	0.8	0.24	<0.001
	P	4.5	0.8	0.17	<0.001
	Microhabitat (shrub)	0.2	0.8	0.01	0.78
	Urban	-2.1	0.9	-0.08	0.02
	Current growing season water availability	54	3	0.54	<0.001
	Previous growing season water availability	-8	5	-0.06	0.09

[1] Taxonomic diversity is defined by species richness and Shannon diversity.

[2] Phylogenetic diversity is defined by SES MPD

APPENDIX C

IRB EXEMPTION FOR HUMAN SUBJECTS RESEARCH

ASU Knowledge Enterprise Development

EXEMPTION GRANTED

Sharon Hall
Life Sciences, School of (SOLS)
480/965-5650
sharonjhall@asu.edu

Dear Sharon Hall:

On 5/1/2018 the ASU IRB reviewed the following protocol:

Type of Review:	Initial Study
Title:	Change over time in yard preferences and actual vegetation in Phoenix residential yards
Investigator:	Sharon Hall
IRB ID:	STUDY00008190
Funding:	Name: ASU: Graduate and Professional Student Association (GPSA)
Grant Title:	
Grant ID:	
Documents Reviewed:	• Initial and Follow-up Contact Postcards, Category: Recruitment Materials; • Contact Letter, Category: Consent Form; • Yard Management Survey Protocol, Category: IRB Protocol; • Yard Management Survey Instrument, Category: Recruitment Materials;

The IRB determined that the protocol is considered exempt pursuant to Federal Regulations 45CFR46 (2) Tests, surveys, interviews, or observation on 5/1/2018.

In conducting this protocol you are required to follow the requirements listed in the INVESTIGATOR MANUAL (HRP-103).

Sincerely,

IRB Administrator

cc: Megan Wheeler
Sharon Hall
Kelli Larson
Megan Wheeler

APPENDIX D

CHAPTER 4 SUPPLEMENTARY TABLES AND FIGURES

Defining the front yard

The front yard was defined as the area between the front sidewalk (or street where there was no sidewalk) and the front of the house (Appendix D: Fig. S1). Where the house was partially set back (i.e., the front of the house was not a straight line), we considered the front yard to end at the front door unless the front door was set back by more than half the length of the driveway (i.e., much of the front yard would be inside the house if drawn as a rectangle). In that case, the front yard was defined as ending at the frontmost part of the house. For corner lots, we considered the front yard to be the part of the lot facing the street matching the house address.

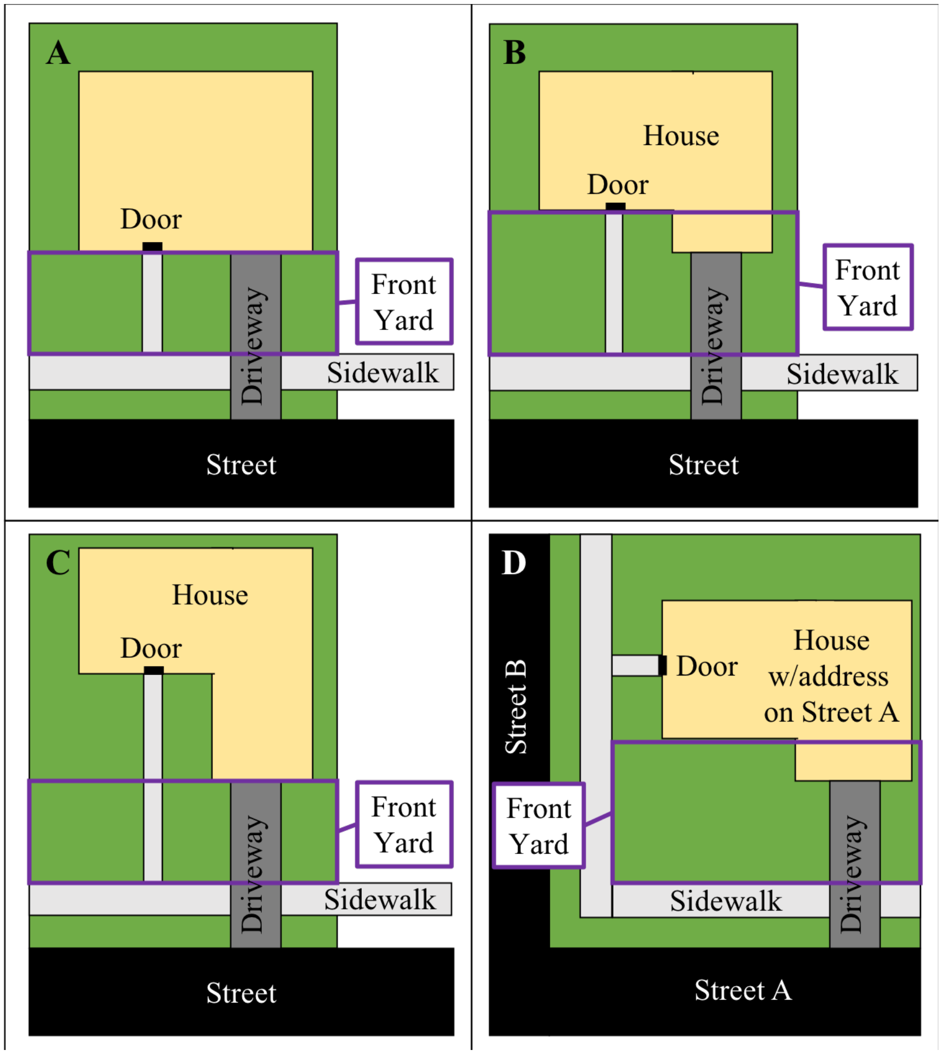

FIG. S1. Definition of front yards for different lot configurations. (A) The front yard was defined as the area between the sidewalk and the front of the house, not including the strip between the sidewalk and street where it existed. (B) If the house was partially set back, the front yard was defined as ending at the front door. (C) If the front door was more than half the length of the driveway back from the furthest front part of the house, then the front yard was defined as ending at the frontmost part of the house. (D) For corner lots, the front yard was considered to be the area facing the street matching the house address.

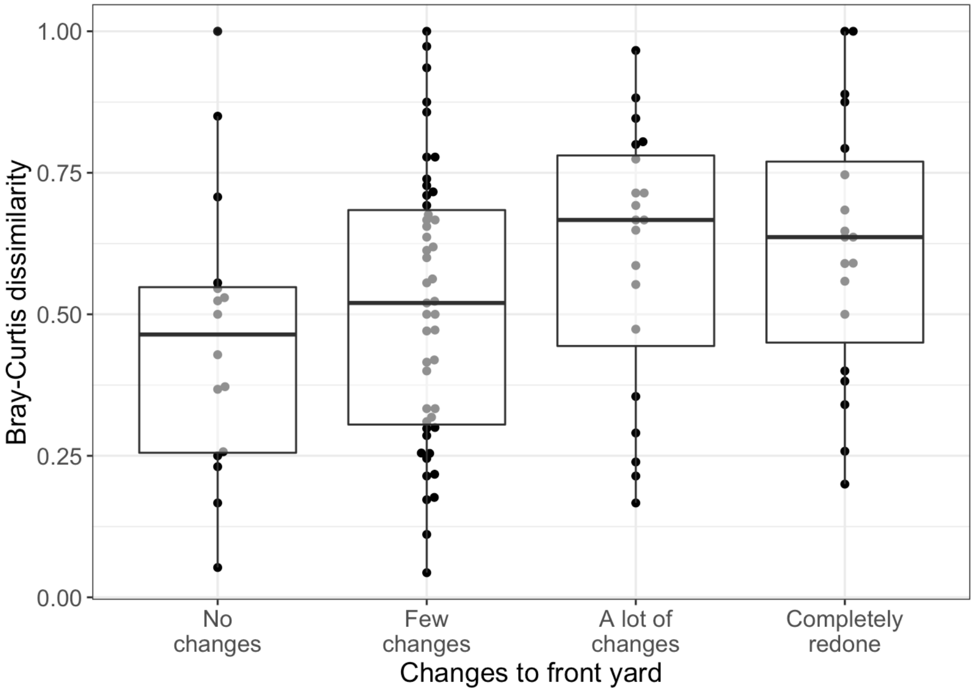

FIG. S2. Resident-reported amount of changes made to the front yard since moving in compared to ten-year vegetation turnover. N = 102.

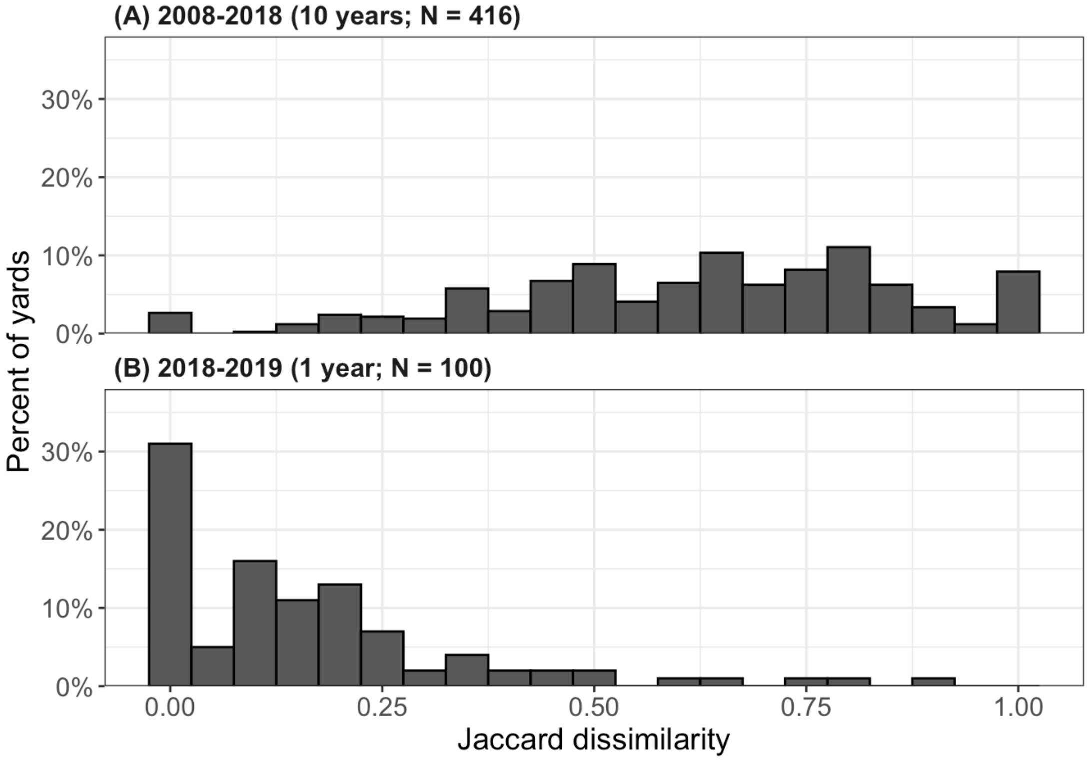

FIG. S3. Jaccard dissimilarity (presence/absence) for front yard woody vegetation at the genus level for 416 yards from 2008 to 2018 and 100 yards from 2018 to 2019.

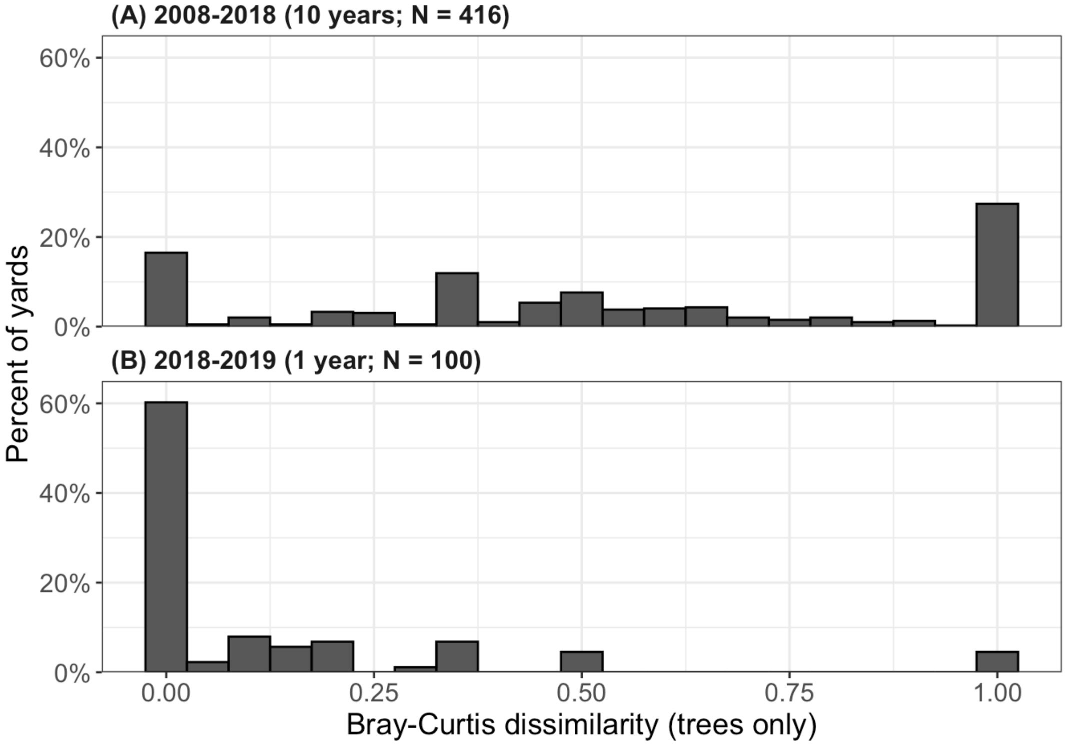

FIG. S4. Bray-Curtis dissimilarity for front yard trees only at the genus level for 416 yards from 2008 to 2018 and 100 yards from 2018 to 2019.

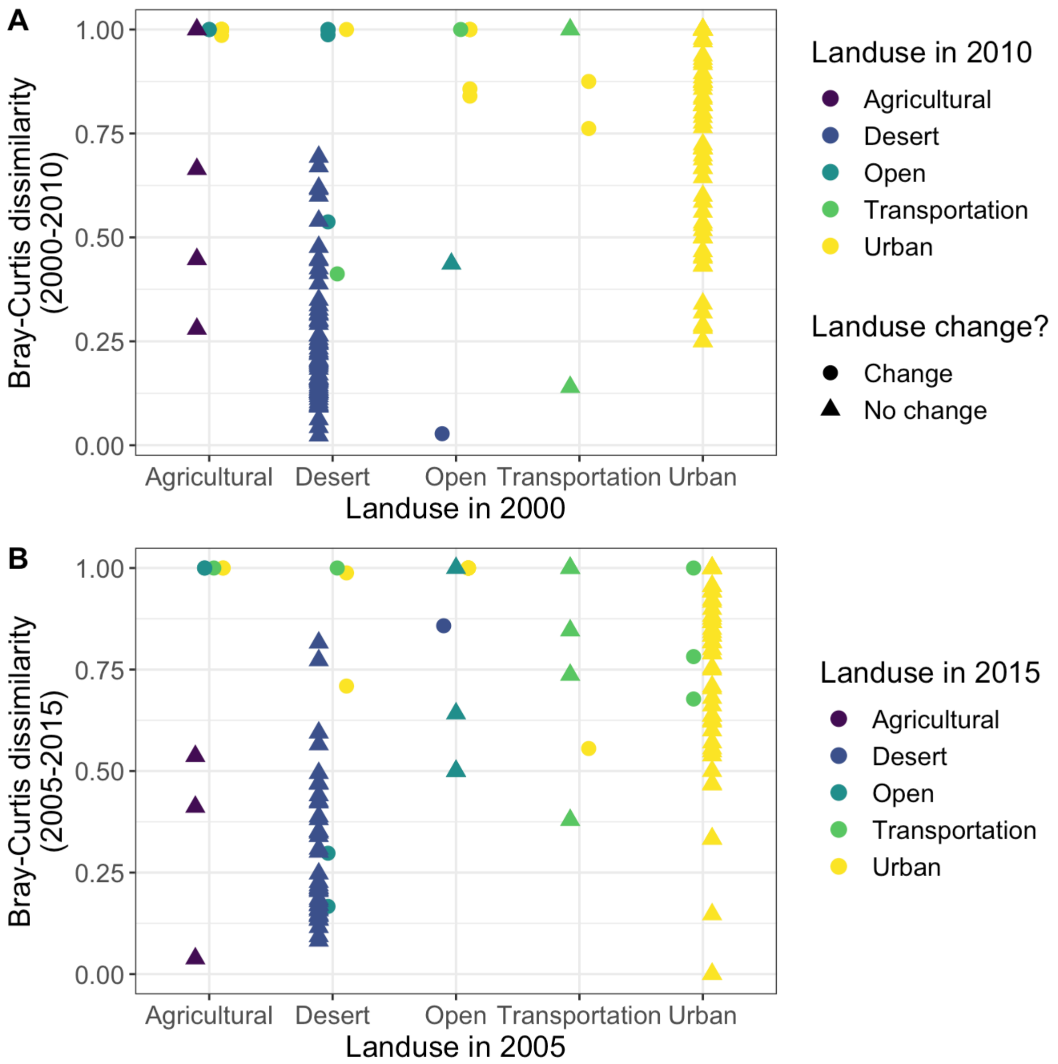

FIG. S5. Woody vegetation community dissimilarity for ESCA plots by dominant land use type in (A) 2000 and 2010; and (B) 2005 and 2015.

TABLE S1. Genera recorded in front yard sidewalk surveys in each year with the percent of sampled yards for that year in which they were recorded. Genera for which multiple growth forms were recorded (e.g., Tree and Shrub growth form) are shown in multiple rows. Total N = 416 for 2008, 416 for 2018, and 100 for 2019.

		% of sampled yards		
Growth form	Genus	2008	2018	2019
Shrub	*Abutilon*	0	<1	0
Tree	*Acacia*	8	5	5
Shrub	*Acca*	0	<1	1
Tree	*Acer*	<1	0	0
Succulent	*Aeonium*	0	<1	0
Succulent	*Agave*	32	38	31
Tree	*Ailanthus*	<1	0	0
Tree	*Albizia*	5	2	1
Succulent	*Aloe*	16	30	18
Shrub	*Alyogyne*	0	<1	2
Tree	*Alyogyne*	0	<1	0
Shrub	*Anigozanthos*	0	<1	0
Vine	*Antigonon*	<1	<1	1
Tree	*Araucaria*	<1	<1	0
Succulent	*Armatocereus*	2	1	1
Shrub	*Artemisia*	1	<1	0
Shrub	*Asclepias*	<1	1	0
Shrub	*Asparagus*	8	8	15
Succulent	*Astroloba*	0	<1	0
Succulent	*Austrocylindropuntia*	0	1	2
Shrub	*Baccharis*	4	2	0
Tree	*Bambusa*	0	<1	0
Shrub	*Bauhinia*	0	<1	0
Tree	*Bauhinia*	4	3	5
Succulent	*Beaucarnea*	0	<1	0
Tree	*Bismarckia*	<1	<1	1
Shrub	*Bougainvillea*	47	29	27
Tree	*Brachychiton*	7	4	4
Tree	*Brahea*	3	<1	1
Tree	*Broussonetia*	<1	<1	1
Succulent	*Bryophyllum*	0	<1	0
Shrub	*Buddleja*	1	1	1
Tree	*Butia*	0	2	2
Shrub	*Buxus*	0	2	2
Shrub	*Caesalpinia*	22	15	9
Tree	*Caesalpinia*	0	2	3
Shrub	*Calliandra*	9	6	8
Shrub	*Callistemon*	6	10	9
Tree	*Callistemon*	4	3	4
Vine	*Campsis*	<1	<1	1
Shrub	*Canna*	8	3	4
Shrub	*Carissa*	25	15	21
Succulent	*Carnegiea*	7	8	12
Tree	*Carya*	<1	<1	1
Shrub	*Cascabela*	4	<1	1
Tree	*Cascabela*	0	3	5

Growth form	Genus	% of sampled yards 2008	 2018	 2019
Succulent	*Cephalocereus*	0	<1	2
Tree	*Ceratonia*	0	<1	0
Succulent	*Cereus*	9	5	6
Tree	*Chamaerops*	7	6	5
Tree	*Chilopsis*	<1	1	3
Tree	*Chitalpa*	0	<1	0
Shrub	*Chrysanthemum*	0	<1	1
Tree	*Citrus*	12	13	11
Shrub	*Convolvulus*	2	1	1
Shrub	*Cordia*	1	0	1
Tree	*Cordia*	<1	0	0
Tree	*Corymbia*	<1	0	0
Succulent	*Crassula*	0	<1	0
Shrub	*Cuphea*	0	<1	2
Shrub	*Cupressus*	0	<1	0
Tree	*Cupressus*	6	2	0
Tree	*Cycas*	7	7	11
Succulent	*Cylindropuntia*	2	1	1
Shrub	*Dalbergia*	0	<1	0
Tree	*Dalbergia*	3	4	5
Succulent	*Dasylirion*	12	9	9
Tree	*Delonix*	0	<1	1
Shrub	*Dipladenia*	0	1	0
Shrub	*Dodonaea*	0	2	4
Tree	*Dodonaea*	0	<1	0
Vine	*Dolichandra*	6	4	1
Shrub	*Dracaena*	0	<1	0
Shrub	*Duranta*	<1	2	3
Tree	*Duranta*	0	<1	1
Shrub	*Ebenopsis*	0	<1	0
Tree	*Ebenopsis*	4	4	4
Succulent	*Echeveria*	0	<1	0
Succulent	*Echinocactus*	12	13	13
Succulent	*Echinocereus*	3	3	2
Succulent	*Echinopsis*	3	4	7
Shrub	*Encelia*	3	3	1
Shrub	*Eremophila*	1	4	4
Shrub	*Ericameria*	0	1	2
Succulent	*Escobaria*	<1	0	0
Shrub	*Eucalyptus*	0	<1	0
Tree	*Eucalyptus*	4	3	5
Shrub	*Euphorbia*	2	6	5
Succulent	*Euphorbia*	1	8	15
Tree	*Eysenhardtia*	<1	<1	0
Succulent	*Ferocactus*	8	10	11
Shrub	*Ficus*	<1	2	2
Tree	*Ficus*	10	8	7
Vine	*Ficus*	3	2	5
Tree	*Fortunella*	<1	<1	1
Succulent	*Fouquieria*	12	11	10
Tree	*Fraxinus*	7	5	6
Shrub	*Galvezia*	0	<1	0

Growth form	Genus	% of sampled yards 2008	 2018	 2019
Shrub	*Gardenia*	<1	1	4
Tree	*Geoffroea*	0	<1	2
Tree	*Gleditsia*	1	<1	0
Shrub	*Grewia*	0	<1	0
Shrub	*Guaiacum*	<1	0	0
Shrub	*Hamelia*	0	<1	1
Tree	*Handroanthus*	0	<1	1
Vine	*Hardenbergia*	<1	<1	0
Succulent	*Harrisia*	0	<1	0
Succulent	*Haworthia*	0	<1	0
Vine	*Hedera*	2	<1	0
Shrub	*Hesperaloe*	25	26	33
Succulent	*Hesperoyucca*	<1	0	0
Shrub	*Hibiscus*	14	12	13
Vine	*Ipomoea*	0	<1	0
Tree	*Jacaranda*	9	6	7
Shrub	*Jasminum*	<1	2	3
Tree	*Juglans*	<1	<1	0
Shrub	*Juniperus*	10	2	2
Tree	*Juniperus*	0	4	2
Shrub	*Justicia*	4	3	3
Succulent	*Kalanchoe*	0	1	2
Tree	*Lagerstroemia*	0	<1	1
Shrub	*Lantana*	40	33	46
Shrub	*Larrea*	<1	1	2
Shrub	*Lavandula*	<1	2	2
Shrub	*Leucophyllum*	45	34	37
Shrub	*Ligustrum*	5	3	5
Tree	*Ligustrum*	0	<1	0
Succulent	*Lophocereus*	6	8	11
Shrub	*Lycianthes*	<1	1	2
Tree	*Lysiloma*	4	7	3
Tree	*Magnolia*	<1	0	0
Shrub	*Malpighia*	<1	0	0
Tree	*Malus*	<1	<1	0
Succulent	*Mammillaria*	2	1	2
Succulent	*Marginatocereus*	4	4	3
Tree	*Mariosousa*	<1	<1	0
Shrub	*Maytenus*	1	<1	1
Tree	*Melia*	3	2	0
Tree	*Morus*	8	4	2
Shrub	*Musa*	3	<1	1
Succulent	*Myrtillocactus*	2	1	3
Shrub	*Myrtus*	15	13	16
Tree	*Myrtus*	0	1	1
Shrub	*Nandina*	4	3	7
Shrub	*Nerium*	31	20	21
Tree	*Nerium*	0	2	4
Shrub	*Olea*	0	1	2
Tree	*Olea*	4	5	8
Tree	*Olneya*	<1	<1	0
Succulent	*Opuntia*	18	18	9

Growth form	Genus	% of sampled yards 2008	 2018	 2019
Succulent	*Pachycereus*	0	2	0
Succulent	*Pachyphytum*	0	<1	0
Shrub	*Pachypodium*	0	<1	0
Tree	*Parkinsonia*	15	14	12
Vine	*Parthenocissus*	0	<1	0
Tree	*Persea*	0	<1	0
Shrub	*Philadelphus*	1	0	0
Tree	*Phoenix*	17	20	23
Tree	*Pinus*	8	5	3
Tree	*Pistacia*	2	2	5
Shrub	*Pittosporum*	7	7	11
Tree	*Pittosporum*	0	<1	0
Tree	*Platanus*	<1	<1	1
Shrub	*Plumbago*	<1	2	1
Shrub	*Plumeria*	<1	<1	1
Tree	*Podocarpus*	0	<1	0
Tree	*Populus*	1	<1	0
Succulent	*Portulaca*	0	<1	0
Succulent	*Portulacaria*	1	9	8
Tree	*Prosopis*	22	19	13
Tree	*Prunus*	3	1	0
Tree	*Psidium*	0	<1	0
Shrub	*Punica*	2	2	2
Tree	*Punica*	0	1	0
Shrub	*Pyracantha*	3	<1	0
Tree	*Pyrus*	0	1	1
Tree	*Quercus*	1	1	2
Shrub	*Rhaphiolepis*	1	2	0
Shrub	*Rhus*	<1	<1	1
Tree	*Rhus*	1	<1	0
Tree	*Robinia*	<1	0	0
Shrub	*Rosa*	17	15	10
Shrub	*Rosmarinus*	12	10	11
Shrub	*Ruellia*	26	20	20
Shrub	*Russelia*	<1	3	8
Shrub	*Salvia*	<1	0	0
Tree	*Sambucus*	<1	0	0
Shrub	*Sansevieria*	1	1	1
Succulent	*Sansevieria*	0	<1	2
Tree	*Sapindus*	1	0	0
Tree	*Schinus*	0	1	1
Shrub	*Searsia*	0	<1	1
Tree	*Searsia*	3	4	0
Succulent	*Sedum*	0	<1	0
Shrub	*Senna*	10	7	10
Shrub	*Simmondsia*	1	1	0
Vine	*Solanum*	<1	0	0
Shrub	*Sophora*	1	<1	0
Tree	*Sophora*	1	2	3
Shrub	*Sphaeralcea*	2	<1	0
Vine	*Sphagneticola*	<1	2	3
Shrub	*Spiraea*	0	<1	1

		% of sampled yards		
Growth form	Genus	2008	2018	2019
Succulent	*Stenocereus*	2	<1	2
Succulent	*Stetsonia*	0	<1	0
Shrub	*Strelitzia*	<1	4	5
Tree	*Syagrus*	11	5	7
Shrub	*Syringa*	<1	0	0
Shrub	*Tecoma*	22	22	24
Tree	*Tecoma*	0	<1	0
Shrub	*Teucrium*	0	<1	2
Tree	*Thuja*	<1	<1	1
Tree	*Tipuana*	0	<1	0
Shrub	*Trachelospermum*	<1	<1	2
Vine	*Trachelospermum*	0	<1	1
Tree	*Trachycarpus*	4	1	1
Tree	*Ulmus*	6	9	7
Shrub	Unidentifiable	20	19	27
Succulent	Unidentifiable	5	8	15
Tree	Unidentifiable	6	4	1
Vine	Unidentifiable	7	3	1
Shrub	*Verbena*	<1	0	0
Vine	*Vigna*	1	0	0
Shrub	*Vitex*	0	<1	1
Tree	*Vitex*	1	3	3
Vine	*Vitis*	0	<1	0
Tree	*Washingtonia*	16	20	20
Shrub	*Xylosma*	0	<1	0
Succulent	*Yucca*	20	14	11
Shrub	*Zauschneria*	<1	0	0
Tree	*Ziziphus*	0	<1	0

TABLE S2. 2018 social survey respondent demographics. One respondent with no ID is not included. % shows the percentage of respondents for a particular question who are in the given category.

	Value	Old Hispanic Core	Historic Palms District	New Xeric Tracts	Wealthy Mountain Oasis	Total
Owners	%	83	95	90	100	94
Female	%	75	71	67	64	68
Bachelor's or graduate degree	%	0	85	63	93	77
White	%	40	97	96	96	94
Income category[1]	mean ± s.d.	1 ± 0.6	9 ± 3	5 ± 2	9 ± 3	8 ± 3
	range	1-2	2-11	2-10	4-11	1-11
Respondent age (years)	mean ± s.d.	53 ± 24	59 ± 15	52 ± 14	55 ± 11	56 ± 15
	range	23-80	30-91	28-82	32-74	23-91
Percent life in Phoenix	mean ± s.d.	95 ± 13	57 ± 28	45 ± 22	34 ± 16	49 ± 27
	range	68-100	0-100	7-100	6-86	0-100
Years in current home	mean ± s.d.	30 ± 21	18 ± 16	14 ± 10	13 ± 8	16 ± 14
	range	3-58	0-56	1-31	1-28	0-58
Home age (years)[2]	mean ± s.d.	65 ± 7	80 ± 7	24 ± 5	9 ± 3	46 ± 25
	range	16-72	58-97	14-33	4-11	14-97

[1] Ordinal variable

[2] From county tax assessor records, not survey responses

TABLE S3. Bray-Curtis and Jaccard dissimilarity for different sample comparisons.

		Bray-Curtis		Jaccard	
	N	Mean	Median	Mean	Median
2008-2018	416	0.58	0.60	0.62	0.67
2018-2019	100	0.18	0.13	0.16	0.12
2008-2018 trees only	416	0.54	0.50	0.57	0.67
2008-2018 no change in owner/typology	216	0.51	0.51	0.56	0.59
2008-2018 change in typology	61	0.61	0.67	0.69	0.71
2008-2018 change in owner	101	0.61	0.63	0.65	0.67
2008-2018 change in owner and typology	37	0.73	0.76	0.78	0.80

APPENDIX E

NATIVITY CLASSIFICATION METHODS

To classify nativity, we first looked up all observed species in the USDA Plants database and the Biota of North America Program database (BONAP; Kartesz 2015, USDA NRCS 2020). If any part of a species' county-level native distribution fell within the U.S. Sonoran Desert regions (La Paz, Yuma, Maricopa, Pinal, and Pima counties), the species was considered 'native' to the Arizona Sonoran Desert and to the Sonoran Desert. Species that were not native to Arizona were compared against a checklist of native vascular plants of Mexico and classified as native to the Sonoran Desert (but not the Arizona Sonoran Desert) if they were native to Baja California Sur, Baja California Norte, or Sonora (Villaseñor, 2016). Many cultivated species were not recorded on USDA Plants or BONAP because they do not grow outside of cultivation in the U.S. We looked up the description and origins of these species using a cultivated plant encyclopedia and classified them as non-native unless they were listed as originating in North America or Mexico (Bailey & Bailey, 1976). If a species did not meet the above criteria for nativity, it was classified as non-native.

For identifications made at the genus level, plants were classified as native if all species of the genus in USDA Plants were native (e.g., *Abutilon* sp., *Limonium* sp.), and as non-native if there were no native species in these databases (e.g., *Eucalyptus* sp., *Rosa* sp.). For cases where both native and non-native species could be found (e.g., *Opuntia* sp., *Prosopis* sp.), we used available nursery and urban forest inventories to categorize genera based on the most likely species observed where possible. For tree genera, we used the Phoenix Urban Forest Assessment (2014) to identify the most common species in the urban forest. We then classified the genus based on the nativity of the most common species (e.g., *Prosopis veluntina* is the most commonly-observed species of *Prosopis* and is native to the Arizona Sonoran Desert, so *Prosopis* sp. is classified as native). Genera containing commonly-cultivated non-native species and rare native congeners (e.g., *Plumbago* sp., *Lantana* sp.) were classified as non-native.

Four genera (*Agave* sp., *Ferocactus* sp., *Optunia* sp., and *Yucca* sp.) were relatively common in the sample, but likely contained a mix of native, non-native, hybrid, and cultivar individuals. We classified these individuals as native to both the Sonoran and the Arizona Sonoran Desert, as there are cultivated species in each of these genera that are native, and many individuals were horticultural varieties without clear nativities. Further, *Leucophyllum* sp. was classified as native to the Sonoran Desert because the most common of the three cultivated species in this genus found in Phoenix (*L. frutescens*) is native to Sonora, Mexico, though not Arizona. The other species (*L. candidum* and *L. laevigatum*) are native to the Chihuahuan Desert in nearby regions of the U.S. and Mexico.

Following these classifications, we addressed nativity of individuals that could not be accurately identified to genus. To avoid considering all unidentified individuals as non-native by default, we classified unidentifiable barrel, hedgehog, and columnar cacti as native, and classified unidentifiable broadleaf trees, broadleaf shrubs, and hedge-type shrubs as non-native. We assume in these classifications that unidentifiable cacti are likely uncommon natives or are functionally similar to natives, while unidentifiable broadleaf trees and shrubs would have been identified if they had been one of the few native species with these growth forms.

With these methods, <1% of individuals remained unclassified due to either the lack of identification to genus or similar likelihood of a genus being either native or non-native (*Acacia* sp. and *Cephalocereus* sp.; Appendix F: Table S1).

REFERENCES

Bailey, L. H., and E. Z. Bailey. 1976. Hortus third: A concise dictionary of plants cultivated in the United States and Canada. Macmillan Publishing Company, New York.

Kartesz, J. T. 2015. North American plant atlas. The Biota of North America Program (BONAP), Chapel Hill, North Carolina, USA. http://bonap.net/napa.

Phoenix, Arizona, Community Forest Assessment. 2014. Davey Resource Group, Phoenix, Arizona, USA.

USDA NRCS. 2020. The PLANTS database. National Plant Data Team, Greensboro, North Carolina, USA. http://plants.usda.gov.

Villaseñor, J. L. 2016. Checklist of the native vascular plants of Mexico. Revista Mexicana de Biodiversidad 87:559–902. 299

APPENDIX F

NATIVITY CLASSIFICATION TABLE

TABLE S1. Species nativity classifications and frequencies.

Species	Growth form	Nativity Arizona Sonoran	Sonoran	Individuals observed	Parcels containing
Abutilon sp.	Shrub	Native	Native	1	1
Acacia constricta	Tree	Native	Native	2	2
Acacia farnesiana	Tree	Native	Native	8	6
Acacia salicina	Tree	Nonnative	Nonnative	4	4
Acacia sp.	Tree	Not classified	Not classified	5	2
Acacia stenophylla	Tree	Nonnative	Nonnative	3	2
Acca sellowiana	Shrub	Nonnative	Nonnative	4	2
Aeonium sp.	Succulent	Nonnative	Nonnative	1	1
Agave americana	Succulent	Native	Native	12	5
Agave attenuata	Succulent	Nonnative	Nonnative	3	1
Agave geminiflora	Succulent	Nonnative	Nonnative	15	10
Agave horrida	Succulent	Nonnative	Nonnative	35	13
Agave lechuguilla	Succulent	Nonnative	Nonnative	3	2
Agave macroacantha	Succulent	Nonnative	Nonnative	4	2
Agave murpheyi	Succulent	Native	Native	4	4
Agave parryi	Succulent	Native	Native	31	11
Agave sp.	Succulent	Native	Native	278	75
Agave victoriae-reginae	Succulent	Nonnative	Nonnative	11	5
Agave vilmoriniana	Succulent	Nonnative	Native	17	2
Agave vivipara	Succulent	Nonnative	Nonnative	1	1
Agave weberi	Succulent	Nonnative	Nonnative	10	4
Albizia julibrissin	Tree	Nonnative	Nonnative	21	8
Aloe sp.	Succulent	Nonnative	Nonnative	413	93
Aloe variegata	Succulent	Nonnative	Nonnative	8	4
Aloe vera	Succulent	Nonnative	Nonnative	72	21
Alyogyne huegelii	Shrub	Nonnative	Nonnative	2	2
Alyogyne huegelii	Tree	Nonnative	Nonnative	4	1
Anigozanthos sp.	Shrub	Nonnative	Nonnative	1	1
Antigonon leptopus	Vine	Nonnative	Native	2	1
Araucaria heterophylla	Tree	Nonnative	Nonnative	3	2
Armatocereus cartwrightianus	Succulent	Nonnative	Nonnative	3	3
Armatocereus laetus	Succulent	Nonnative	Nonnative	2	2
Artemisia schmidtiana	Shrub	Nonnative	Nonnative	3	1
Asclepias sp.	Shrub	Native	Native	18	2
Asclepias subulata	Shrub	Native	Native	2	2
Asparagus densiflorus	Shrub	Nonnative	Nonnative	64	35
Astroloba sp.	Succulent	Nonnative	Nonnative	1	1
Austrocylindropuntia subulata	Succulent	Nonnative	Nonnative	12	6

		Nativity			
Species	Growth form	Arizona Sonoran	Sonoran	Individuals observed	Parcels containing
Baccharis sarothroides	Shrub	Native	Native	10	7
Bambusa sp.	Tree	Nonnative	Nonnative	1	1
Bauhinia divaricata	Shrub	Nonnative	Nonnative	1	1
Bauhinia purpurea	Tree	Nonnative	Nonnative	17	12
Bauhinia sp.	Shrub	Nonnative	Nonnative	1	1
Bauhinia variegata	Tree	Nonnative	Nonnative	1	1
Beaucarnea recurvata	Succulent	Nonnative	Nonnative	1	1
Bismarckia nobilis	Tree	Nonnative	Nonnative	2	2
Bougainvillea sp.	Shrub	Nonnative	Nonnative	249	120
Brachychiton populneus	Tree	Nonnative	Nonnative	17	15
Brachychiton sp.	Tree	Nonnative	Nonnative	1	1
Brahea armata	Tree	Nonnative	Native	3	2
Broussonetia papyrifera	Tree	Nonnative	Nonnative	1	1
Bryophyllum daigremontianum	Succulent	Nonnative	Nonnative	2	1
Bryophyllum sp.	Succulent	Nonnative	Nonnative	6	2
Buddleja marrubiifolia	Shrub	Nonnative	Nonnative	3	3
Buddleja sp.	Shrub	Nonnative	Nonnative	1	1
Butia capitata	Tree	Nonnative	Nonnative	9	8
Buxus microphylla	Shrub	Nonnative	Nonnative	67	7
Caesalpinia gilliesii	Shrub	Nonnative	Nonnative	16	11
Caesalpinia gilliesii	Tree	Nonnative	Nonnative	4	4
Caesalpinia mexicana	Shrub	Nonnative	Nonnative	10	6
Caesalpinia mexicana	Tree	Nonnative	Nonnative	6	2
Caesalpinia pulcherrima	Shrub	Nonnative	Nonnative	56	36
Caesalpinia pulcherrima	Tree	Nonnative	Nonnative	1	1
Caesalpinia sp.	Shrub	Nonnative	Nonnative	12	8
Calliandra californica	Shrub	Nonnative	Native	41	21
Calliandra eriophylla	Shrub	Native	Native	7	2
Calliandra sp.	Shrub	Native	Native	4	3
Callistemon citrinus	Shrub	Nonnative	Nonnative	97	40
Callistemon rigidus	Tree	Nonnative	Nonnative	1	1
Callistemon sp.	Shrub	Nonnative	Nonnative	2	2
Callistemon viminalis	Tree	Nonnative	Nonnative	11	8
Campsis radicans	Vine	Nonnative	Nonnative	3	2
Canna generalis	Shrub	Nonnative	Nonnative	84	12

Species	Growth form	Nativity Arizona Sonoran	 Sonoran	Individuals observed	Parcels containing
Carissa macrocarpa	Shrub	Nonnative	Nonnative	235	62
Carnegiea gigantea	Succulent	Native	Native	47	33
Carya illinoinensis	Tree	Native	Native	1	1
Cascabela thevetia	Shrub	Nonnative	Nonnative	5	3
Cascabela thevetia	Tree	Nonnative	Nonnative	13	11
Cephalocereus sp.	Succulent	Not classified	Not classified	3	3
Ceratonia siliqua	Tree	Nonnative	Nonnative	4	3
Cereus hildmannianus	Succulent	Nonnative	Nonnative	9	9
Cereus repandus	Succulent	Nonnative	Nonnative	1	1
Cereus sp.	Succulent	Nonnative	Nonnative	15	9
Chamaerops humilis	Tree	Nonnative	Nonnative	41	25
Chilopsis linearis	Tree	Native	Native	6	5
Chitalpa tashkentensis	Tree	Nonnative	Nonnative	1	1
Chrysanthemum indicum	Shrub	Nonnative	Nonnative	9	2
Citrus aurantiifolia	Tree	Nonnative	Nonnative	21	13
Citrus aurantium	Tree	Nonnative	Nonnative	15	9
Citrus limon	Tree	Nonnative	Nonnative	1	1
Citrus paradisi	Tree	Nonnative	Nonnative	1	1
Citrus reticulata	Tree	Nonnative	Nonnative	1	1
Citrus sinensis	Tree	Nonnative	Nonnative	18	5
Citrus sp.	Tree	Nonnative	Nonnative	29	19
Citrus trifoliata	Tree	Nonnative	Nonnative	1	1
Convolvulus cneorum	Shrub	Nonnative	Nonnative	13	4
Crassula sp.	Succulent	Nonnative	Nonnative	3	2
Crassula tetragona	Succulent	Nonnative	Nonnative	1	1
Cuphea hyssopifolia	Shrub	Nonnative	Native	7	3
Cupressus arizonica	Shrub	Native	Native	1	1
Cupressus sempervirens	Tree	Nonnative	Nonnative	13	10
Cycas revoluta	Tree	Nonnative	Nonnative	52	29
Cylindropuntia arbuscula	Succulent	Native	Native	1	1
Cylindropuntia sp.	Succulent	Native	Native	3	3
Dalbergia sissoo	Shrub	Nonnative	Nonnative	1	1
Dalbergia sissoo	Tree	Nonnative	Nonnative	25	18
Dasylirion acrotrichum	Succulent	Nonnative	Nonnative	3	2
Dasylirion quadrangulatum	Succulent	Nonnative	Nonnative	4	2
Dasylirion wheeleri	Succulent	Native	Native	51	34
Delonix regia	Tree	Nonnative	Nonnative	1	1
Dipladenia sp.	Shrub	Nonnative	Nonnative	7	4
Dodonaea viscosa	Shrub	Native	Native	29	10

Species	Growth form	Nativity Arizona Sonoran	Nativity Sonoran	Individuals observed	Parcels containing
Dodonaea viscosa	Tree	Native	Native	1	1
Dolichandra unguis-cati	Vine	Nonnative	Nonnative	17	16
Dracaena sp.	Shrub	Nonnative	Nonnative	2	2
Duranta erecta	Shrub	Native	Native	24	10
Duranta erecta	Tree	Native	Native	1	1
Ebenopsis ebano	Shrub	Nonnative	Nonnative	6	1
Ebenopsis ebano	Tree	Nonnative	Nonnative	15	15
Echeveria sp.	Succulent	Nonnative	Nonnative	4	1
Echinocactus grusonii	Succulent	Nonnative	Nonnative	116	53
Echinocactus sp.	Succulent	Native	Native	1	1
Echinocereus engelmannii	Succulent	Native	Native	8	6
Echinocereus sp.	Succulent	Native	Native	8	6
Echinopsis huascha	Succulent	Nonnative	Nonnative	7	3
Echinopsis oxygona	Succulent	Nonnative	Nonnative	1	1
Echinopsis sp.	Succulent	Nonnative	Nonnative	14	8
Echinopsis terscheckii	Succulent	Nonnative	Nonnative	3	3
Encelia farinosa	Shrub	Native	Native	33	14
Eremophila divaricata	Shrub	Nonnative	Nonnative	1	1
Eremophila hygrophana	Shrub	Nonnative	Nonnative	4	2
Eremophila maculata	Shrub	Nonnative	Nonnative	32	12
Eremophila sp.	Shrub	Nonnative	Nonnative	1	1
Ericameria laricifolia	Shrub	Native	Native	9	5
Eucalyptus erythrocorys	Tree	Nonnative	Nonnative	1	1
Eucalyptus kruseana	Shrub	Nonnative	Nonnative	1	1
Eucalyptus kruseana	Tree	Nonnative	Nonnative	1	1
Eucalyptus microtheca	Tree	Nonnative	Nonnative	1	1
Eucalyptus polyanthemos	Tree	Nonnative	Nonnative	2	2
Eucalyptus sp.	Tree	Nonnative	Nonnative	4	4
Euphorbia antisyphilitica	Succulent	Native	Native	12	4
Euphorbia bracteata	Succulent	Nonnative	Native	2	2
Euphorbia lomelii	Succulent	Nonnative	Native	32	13
Euphorbia milii	Shrub	Nonnative	Nonnative	5	3
Euphorbia resinifera	Succulent	Nonnative	Nonnative	13	11
Euphorbia rigida	Shrub	Nonnative	Nonnative	3	1
Euphorbia sp.	Succulent	Nonnative	Nonnative	3	2
Euphorbia tirucalli	Shrub	Nonnative	Nonnative	41	20

Species	Growth form	Nativity Arizona Sonoran	 Sonoran	Individuals observed	Parcels containing
Euphorbia triangularis	Succulent	Nonnative	Nonnative	5	3
Ferocactus cylindraceus	Succulent	Native	Native	7	4
Ferocactus pilosus	Succulent	Nonnative	Nonnative	2	1
Ferocactus sp.	Succulent	Native	Native	74	31
Ficus elastica	Shrub	Nonnative	Nonnative	6	5
Ficus elastica	Tree	Nonnative	Nonnative	1	1
Ficus pumila	Vine	Nonnative	Nonnative	32	10
Ficus sp.	Shrub	Nonnative	Nonnative	3	3
Ficus sp.	Tree	Nonnative	Nonnative	33	26
Fortunella sp.	Tree	Nonnative	Nonnative	4	3
Fouquieria splendens	Succulent	Native	Native	65	44
Fraxinus anomala	Tree	Native	Native	4	2
Fraxinus sp.	Tree	Native	Native	12	10
Fraxinus velutina	Tree	Native	Native	10	6
Galvezia juncea	Shrub	Nonnative	Native	2	1
Gardenia jasminoides	Shrub	Nonnative	Nonnative	8	4
Geoffroea decorticans	Tree	Nonnative	Nonnative	2	2
Gleditsia triacanthos	Tree	Native	Native	4	2
Grewia occidentalis	Shrub	Nonnative	Nonnative	7	2
Hamelia patens	Shrub	Nonnative	Nonnative	3	2
Handroanthus impetiginosus	Tree	Nonnative	Native	1	1
Hardenbergia violacea	Vine	Nonnative	Nonnative	1	1
Harrisia sp.	Succulent	Nonnative	Nonnative	1	1
Haworthia sp.	Succulent	Nonnative	Nonnative	1	1
Hedera helix	Vine	Nonnative	Nonnative	3	3
Hesperaloe funifera	Shrub	Nonnative	Nonnative	16	6
Hesperaloe parviflora	Shrub	Nonnative	Native	243	103
Hibiscus rosa-sinensis	Shrub	Nonnative	Nonnative	68	34
Hibiscus sp.	Shrub	Nonnative	Nonnative	23	14
Ipomoea purpurea	Vine	Nonnative	Nonnative	3	1
Jacaranda mimosifolia	Tree	Nonnative	Nonnative	23	20
Jasminum sambac	Shrub	Nonnative	Nonnative	23	10
Juglans major	Tree	Native	Native	1	1
Juniperus chinensis	Shrub	Nonnative	Nonnative	12	8
Juniperus chinensis	Tree	Nonnative	Nonnative	7	5
Juniperus monosperma	Shrub	Native	Native	1	1

		Nativity			
Species	Growth form	Arizona Sonoran	Sonoran	Individuals observed	Parcels containing
Juniperus monosperma	Tree	Native	Native	8	8
Juniperus osteosperma	Shrub	Native	Native	2	1
Juniperus osteosperma	Tree	Native	Native	2	1
Justicia californica	Shrub	Native	Native	8	4
Justicia candicans	Shrub	Native	Native	4	3
Justicia sp.	Shrub	Native	Native	1	1
Justicia spicigera	Shrub	Nonnative	Native	12	5
Kalanchoe blossfeldiana	Succulent	Nonnative	Nonnative	3	2
Kalanchoe luciae	Succulent	Nonnative	Nonnative	1	1
Kalanchoe sp.	Succulent	Nonnative	Nonnative	1	1
Kalanchoe tomentosa	Succulent	Nonnative	Nonnative	1	1
Lagerstroemia indica	Tree	Nonnative	Nonnative	3	2
Lantana sp.	Shrub	Nonnative	Nonnative	503	136
Larrea tridentata	Shrub	Native	Native	15	5
Lavandula stoechas	Shrub	Nonnative	Nonnative	14	7
Leucophyllum sp.	Shrub	Nonnative	Native	358	127
Ligustrum japonicum	Shrub	Nonnative	Nonnative	50	11
Ligustrum japonicum	Tree	Nonnative	Nonnative	4	1
Lophocereus schottii	Succulent	Native	Native	44	26
Lycianthes rantonnetii	Shrub	Nonnative	Nonnative	7	4
Lysiloma sp.	Tree	Native	Native	2	2
Lysiloma watsonii	Tree	Native	Native	38	18
Malus domestica	Tree	Nonnative	Nonnative	1	1
Mammillaria sp.	Succulent	Native	Native	5	3
Marginatocereus marginatus	Succulent	Nonnative	Nonnative	22	17
Mariosousa willardiana	Tree	Nonnative	Native	2	1
Maytenus phyllanthoides	Shrub	Nonnative	Native	13	2
Melia azedarach	Tree	Nonnative	Nonnative	16	8
Morus alba	Tree	Nonnative	Nonnative	18	16
Musa sp.	Shrub	Nonnative	Nonnative	13	2
Myrtillocactus geometrizans	Succulent	Nonnative	Nonnative	10	4
Myrtus communis	Shrub	Nonnative	Nonnative	293	55
Myrtus communis	Tree	Nonnative	Nonnative	4	4
Nandina domestica	Shrub	Nonnative	Nonnative	15	12
Nerium oleander	Shrub	Nonnative	Nonnative	263	81
Nerium oleander	Tree	Nonnative	Nonnative	8	8
Olea europaea	Shrub	Nonnative	Nonnative	9	4

Species	Growth form	Nativity Arizona Sonoran	 Sonoran	Individuals observed	Parcels containing
Olea europaea	Tree	Nonnative	Nonnative	23	17
Olneya tesota	Tree	Native	Native	2	1
Opuntia articulata	Succulent	Nonnative	Nonnative	9	3
Opuntia basilaris	Succulent	Native	Native	11	8
Opuntia engelmannii	Succulent	Native	Native	3	3
Opuntia erinacea	Succulent	Native	Native	1	1
Opuntia ficus-indica	Succulent	Nonnative	Native	53	19
Opuntia macrocentra	Succulent	Native	Native	1	1
Opuntia microdasys	Succulent	Nonnative	Native	18	11
Opuntia santarita	Succulent	Native	Native	11	10
Opuntia schickendantzii	Succulent	Nonnative	Nonnative	6	2
Opuntia sp.	Succulent	Native	Native	20	14
Pachycereus pringlei	Succulent	Nonnative	Native	2	2
Pachycereus sp.	Succulent	Native	Native	15	8
Pachyphytum oviferum	Succulent	Nonnative	Nonnative	1	1
Pachypodium sp.	Shrub	Nonnative	Nonnative	3	3
Parkinsonia aculeata	Tree	Native	Native	5	4
Parkinsonia florida	Tree	Native	Native	5	5
Parkinsonia microphylla	Tree	Native	Native	12	11
Parkinsonia praecox	Tree	Native	Native	29	20
Parkinsonia sp.	Tree	Native	Native	12	12
Parthenocissus quinquefolia	Vine	Native	Native	2	2
Persea americana	Tree	Nonnative	Native	1	1
Phoenix canariensis	Tree	Nonnative	Nonnative	22	14
Phoenix dactylifera	Tree	Nonnative	Nonnative	3	3
Phoenix roebelenii	Tree	Nonnative	Nonnative	120	62
Phoenix sp.	Tree	Nonnative	Nonnative	2	2
Pinus canariensis	Tree	Nonnative	Nonnative	3	3
Pinus halepensis	Tree	Nonnative	Nonnative	10	9
Pinus pinea	Tree	Nonnative	Nonnative	3	3
Pinus sp.	Tree	Nonnative	Nonnative	1	1
Pistacia atlantica	Tree	Nonnative	Nonnative	3	3
Pistacia chinensis	Tree	Nonnative	Nonnative	8	7
Pittosporum tobira	Shrub	Nonnative	Nonnative	89	27
Pittosporum tobira	Tree	Nonnative	Nonnative	1	1
Platanus wrightii	Tree	Native	Native	2	2
Plumbago auriculata	Shrub	Nonnative	Nonnative	13	5
Plumbago sp.	Shrub	Nonnative	Nonnative	2	1
Plumeria sp.	Shrub	Nonnative	Nonnative	1	1
Podocarpus sp.	Tree	Nonnative	Nonnative	1	1
Populus nigra	Tree	Nonnative	Nonnative	1	1

Species	Growth form	Nativity Arizona Sonoran	Sonoran	Individuals observed	Parcels containing
Portulaca grandiflora	Succulent	Nonnative	Nonnative	50	1
Portulacaria afra	Succulent	Nonnative	Nonnative	87	38
Prosopis alba	Tree	Nonnative	Nonnative	1	1
Prosopis chilensis	Tree	Nonnative	Nonnative	30	25
Prosopis glandulosa	Tree	Native	Native	18	8
Prosopis sp.	Tree	Native	Native	31	18
Prosopis velutina	Tree	Native	Native	10	8
Prunus cerasifera	Tree	Nonnative	Nonnative	4	4
Prunus persica	Tree	Nonnative	Nonnative	3	1
Psidium guajava	Tree	Nonnative	Nonnative	1	1
Punica granatum	Shrub	Nonnative	Nonnative	14	9
Punica granatum	Tree	Nonnative	Nonnative	5	4
Pyracantha coccinea	Shrub	Nonnative	Nonnative	2	1
Pyrus calleryana	Tree	Nonnative	Nonnative	3	3
Pyrus sp.	Tree	Nonnative	Nonnative	1	1
Quercus arizonica	Tree	Native	Native	1	1
Quercus emoryi	Tree	Native	Native	2	2
Quercus sp.	Tree	Nonnative	Nonnative	1	1
Quercus suber	Tree	Nonnative	Nonnative	1	1
Rhaphiolepis indica	Shrub	Nonnative	Nonnative	27	7
Rhus ovata	Shrub	Native	Native	7	1
Rhus virens	Tree	Native	Native	2	1
Rosa banksiae	Shrub	Nonnative	Nonnative	14	3
Rosa sp.	Shrub	Nonnative	Nonnative	241	59
Rosmarinus officinalis	Shrub	Nonnative	Nonnative	116	40
Ruellia peninsularis	Shrub	Nonnative	Native	79	24
Ruellia simplex	Shrub	Nonnative	Nonnative	385	58
Russelia equisetiformis	Shrub	Nonnative	Nonnative	56	13
Sansevieria cylindrica	Succulent	Nonnative	Nonnative	9	1
Sansevieria trifasciata	Shrub	Nonnative	Nonnative	14	5
Schinus terebinthifolia	Tree	Nonnative	Nonnative	7	5
Searsia lancea	Shrub	Nonnative	Nonnative	4	3
Searsia lancea	Tree	Nonnative	Nonnative	19	13
Sedum nussbaumerianum	Succulent	Nonnative	Nonnative	3	1
Sedum sp.	Succulent	Nonnative	Nonnative	2	1
Senna artemisioides	Shrub	Nonnative	Nonnative	64	23
Senna sp.	Shrub	Nonnative	Nonnative	3	1
Simmondsia chinensis	Shrub	Native	Native	12	5

		Nativity			
Species	Growth form	Arizona Sonoran	Sonoran	Individuals observed	Parcels containing
Sophora secundiflora	Shrub	Nonnative	Nonnative	1	1
Sophora secundiflora	Tree	Nonnative	Nonnative	8	7
Sphaeralcea ambigua	Shrub	Native	Native	5	3
Sphagneticola trilobata	Vine	Nonnative	Nonnative	25	7
Spiraea japonica	Shrub	Nonnative	Nonnative	3	2
Stenocereus sp.	Succulent	Native	Native	2	1
Stenocereus thurberi	Succulent	Native	Native	2	2
Stetsonia coryne	Succulent	Nonnative	Nonnative	5	3
Strelitzia nicolai	Shrub	Nonnative	Nonnative	3	3
Strelitzia reginae	Shrub	Nonnative	Nonnative	20	14
Syagrus romanzoffiana	Tree	Nonnative	Nonnative	27	19
Tecoma capensis	Shrub	Nonnative	Nonnative	75	39
Tecoma stans	Shrub	Native	Native	104	47
Tecoma stans	Tree	Native	Native	2	1
Teucrium fruticans	Shrub	Nonnative	Nonnative	6	3
Thuja plicata	Tree	Nonnative	Nonnative	1	1
Tipuana tipu	Tree	Nonnative	Nonnative	4	3
Trachelospermum jasminoides	Shrub	Nonnative	Nonnative	3	2
Trachelospermum jasminoides	Vine	Nonnative	Nonnative	2	2
Trachycarpus fortunei	Tree	Nonnative	Nonnative	7	5
Ulmus parvifolia	Tree	Nonnative	Nonnative	53	35
Unidentifiable barrel cactus	Succulent	Native	Native	4	2
Unidentifiable broadleaf shrub	Shrub	Nonnative	Nonnative	147	49
Unidentifiable broadleaf tree	Tree	Nonnative	Nonnative	20	11
Unidentifiable columnar cactus	Succulent	Native	Native	33	22
Unidentifiable hedge shrub	Shrub	Nonnative	Nonnative	3	3
Unidentifiable hedgehog cactus	Succulent	Native	Native	4	3
Unidentifiable little leaf shrub	Shrub	Not classified	Not classified	29	14
Unidentifiable little leaf tree	Tree	Not classified	Not classified	4	4
Unidentifiable succulent	Succulent	Not classified	Not classified	6	4
Unidentifiable vine	Vine	Not classified	Not classified	20	12
Vitex agnus-castus	Tree	Nonnative	Nonnative	15	9

		Nativity			
Species	Growth form	Arizona Sonoran	Sonoran	Individuals observed	Parcels containing
Vitex trifolia	Shrub	Nonnative	Nonnative	3	1
Vitex trifolia	Tree	Nonnative	Nonnative	1	1
Vitis vinifera	Vine	Nonnative	Nonnative	1	1
Washingtonia sp.	Tree	Native	Native	163	78
Xylosma sp.	Shrub	Nonnative	Nonnative	11	3
Yucca aloifolia	Succulent	Nonnative	Nonnative	3	2
Yucca elata	Succulent	Native	Native	7	5
Yucca gigantea	Succulent	Nonnative	Nonnative	3	3
Yucca gloriosa	Succulent	Nonnative	Nonnative	19	10
Yucca rigida	Succulent	Nonnative	Native	1	1
Yucca schottii	Succulent	Native	Native	7	2
Yucca sp.	Succulent	Native	Native	56	34
Ziziphus jujuba	Tree	Nonnative	Nonnative	6	1