Ecology and the City: A Long-Term Social-Ecological Examination of the Drivers and Diversity of Urban Vegetation

by

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ABSTRACT

Often, when thinking of cities we envision designed landscapes, where people regulate everything from water to weeds, ultimately resulting in an ecosystem decoupled from biophysical processes. It is unclear, however, what happens when the people regulating these extensively managed landscapes come under stress, whether from unexpected economic fluctuations or from changing climate norms. The overarching question of my dissertation research was: How does urban vegetation change in response to human behavior? To answer this question, I conducted multiscale research in an arid urban ecosystem as well as in a virtual desert city. I used a combination of long-term data and agent-based modeling to examine changes in vegetation across a range of measures influenced by biophysical, climate, institutional, and socioeconomic drivers. At the regional scale, total plant species diversity increased from 2000 to 2010, while species composition became increasingly homogeneous in urban and agricultural areas. At the residential scale, I investigated the effects of biophysical and socioeconomic drivers – the Great Recession of 2007-2010 in particular – on changing residential yard vegetation in Phoenix, AZ. Socioeconomic drivers affected plant composition and increasing richness, but the housing boom from 2000 through 2005 had a stronger influence on vegetation change than the subsequent recession. Surprisingly, annual plant species remained coupled to winter precipitation despite my expectation that their dynamics might be driven by socioeconomic fluctuations. In a modeling experiment, I examined the relative strength of psychological, social, and governance influences on large-scale urban land cover in a desert city. Model results suggested that social norms may be strong enough to lead to large-scale conversion to low water use residential landscaping, and governance may be unnecessary to catalyze residential landscape conversion under the pressure of extreme drought conditions. Overall, my dissertation research showed that urban vegetation is dynamic, even under the presumably stabilizing influence of human management activities. Increasing climate pressure, unexpected socioeconomic disturbances, growing urban populations, and shifting policies all contribute to urban vegetation dynamics. Incorporating these findings into planning policies will contribute to the sustainable management of urban ecosystems.

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INTRODUCTION

As people move into urban areas in greater numbers and more rapidly than ever, there is a growing need to understand the complex relationships between humans and biodiversity in urban landscapes. The interaction of anthropogenic and biophysical processes in urban areas results in spatially heterogeneous land cover. However, studies of urban biotic homogenization show increasing similarity among cities despite their otherwise disparate natural surroundings (McKinney 2006, Trentanovi et al. 2013). Spatial urbanization patterns are also evident in land fragmentation (Shrestha et al. 2012), the urban heat island phenomenon (Buyantuyev and Wu 2009), and primary productivity (Buyantuyev and Wu 2010). Temporal dynamics of urban vegetation have received less attention than these other topics. By increasing our knowledge of the effects of urbanization on urban vegetation this dissertation lays the foundation for researchers to assess urbanization effects on biodiversity, carbon sequestration, water quality maintenance, and other urban ecosystem services important to human well-being and mediated by vegetation (Peterson et al. 2010, Eigenbrod et al. 2011). This research is directly relevant to arid cities, but more broadly, the approach and conceptual framework are relevant to studies of various urban ecosystems (Fig. 0.1).

In natural systems, events like drought or insect outbreaks are key forms of exogenous disturbance that help structure plant communities (Pickett and White 1985). But in a social-ecological system such as the Phoenix, Arizona, USA metropolitan area natural disturbances are often mitigated by human actions including supplemental watering and the use of pesticides. In urban systems, the disturbance of interest may originate not from exogenous natural forces but instead from socioeconomic forces. Between 2000 and 2006, the average U.S. house price increased by more than 50%, with the peak increase occurring in 2004 (Schluter et al. 2012). The subsequent housing bubble collapse, despite the existence of economic disturbance precedents (e.g. the stock market bubble of the late 1990s), shocked the U.S. economy. Economists acknowledge that the Great Recession, despite its origin in the U.S. real estate market, further expanded to affect financial markets around the world (Allen et al. 2009). Financial crises often result from real estate boom-bust cycles, and are followed by several years of rising unemployment and falling home values (Reinhart and Rogoff 2008, 2009). The Great Recession spread to global financial markets, increased U.S. unemployment rates, and decreased home equity, and it is conceivable that the socioeconomic shock also resulted in changes to land cover and biodiversity (Prishchepov et al. 2012), especially in urban areas.

STRUCTURE OF THE DISSERTATION

In Chapter 1, I examine pre- and post-recession landscapes across a gradient of human influence. In urban social-ecological systems, human management activities like supplemental watering and the use of pesticides mitigate most natural disturbances, and result in a highly managed and relatively stable urban landscapes (Knapp et al. 2012). Landscape design and management continually structure the diversity and composition of urban plant communities, but a sudden unexpected shock like the Great Recession acts as a socioeconomic disturbance that affects urban vegetation across scales (Fig. 0.1, Arrow 3). I ask whether plant diversity and biotic homogenization increased following the Great

Recession. I also examined how designed urban landscapes and their surrounding desert and agricultural counterparts varied in plant diversity and community composition (Fig. 0.1, Arrows 1 & 2). This work was completed in collaboration with Janet Franklin (School of Geographical Sciences and Urban Planning, Arizona State University) and Scott Collins (Department of Biology, University of New Mexico).

In Chapter 2, I explore the socioeconomic and biophysical drivers of residential vegetation, using Phoenix, AZ as a case study (Fig. 0.1, Boxes B, C, E). Defining socioeconomic disturbance as *a profound change to a social-ecological system that is caused by a relatively abrupt disruption to economic activity*, I analyze drivers of residential vegetation richness and composition before, during, and after the Great Recession (Fig. 0.1, Box E). I ask whether the strength of socioeconomic factors changes with socioeconomic disturbance and examine the role of biophysical drivers in heavily managed residential areas. I expected that the Great Recession would indirectly release residential landscapes from human controls, and result in an increase in annual and weedy early successional species. This work was completed in collaboration with Janet Franklin (School of Geographical Sciences and Urban Planning, Arizona State University), Scott Collins (Department of Biology, University of New Mexico), and Abigail York (School of Human Evolution and Social Change, Arizona State University).

In Chapter 3, I explore the role human decisions play on large-scale vegetation cover under the stress of climate change (Fig. 0.1, Arrow 4). People’s decisions are influenced by psychological, social, and economic stimuli. To quantitatively examine the relative strength and interaction of these influences, I present the results of an agent-based model experiment. I ask about the strength of cultural norms in achieving large-scale landscaping change (Fig. 0.1, Box D), whether adding formal institutional rules with penalties improves the results, and examine the effects of population density on the resulting scenarios. This work was completed in collaboration with Abigail York (School of Human Evolution and Social Change, Arizona State University).

Finally, in the last chapter I provide a brief synthesis of the overall findings of this research. I also suggest some next steps and summarize the major contributions.

PRODUCTS OF DISSERTATION CHAPTERS

Chapter 1

Ripplinger, J., J. Franklin, S.L. Collins. *In review*. When the economic engine stalls – A multi-scale comparison of vegetation dynamics in pre- and post-recession Phoenix, Arizona, USA. *Landscape and Urban Planning*

Chapter 2

Ripplinger, J, J. Franklin, S.L. Collins, A.M. York. *In prep.* Boom-bust economics and the ecology of cities: How strong is the link? For *Frontiers in Ecology and the Environment*

Chapter 3

Ripplinger, J. A.M. York. *In prep*. Residential landscaping shifts under climate change: Are cultural norms enough to trigger large-scale transitions to low water use vegetation? For *Ecology and Society*

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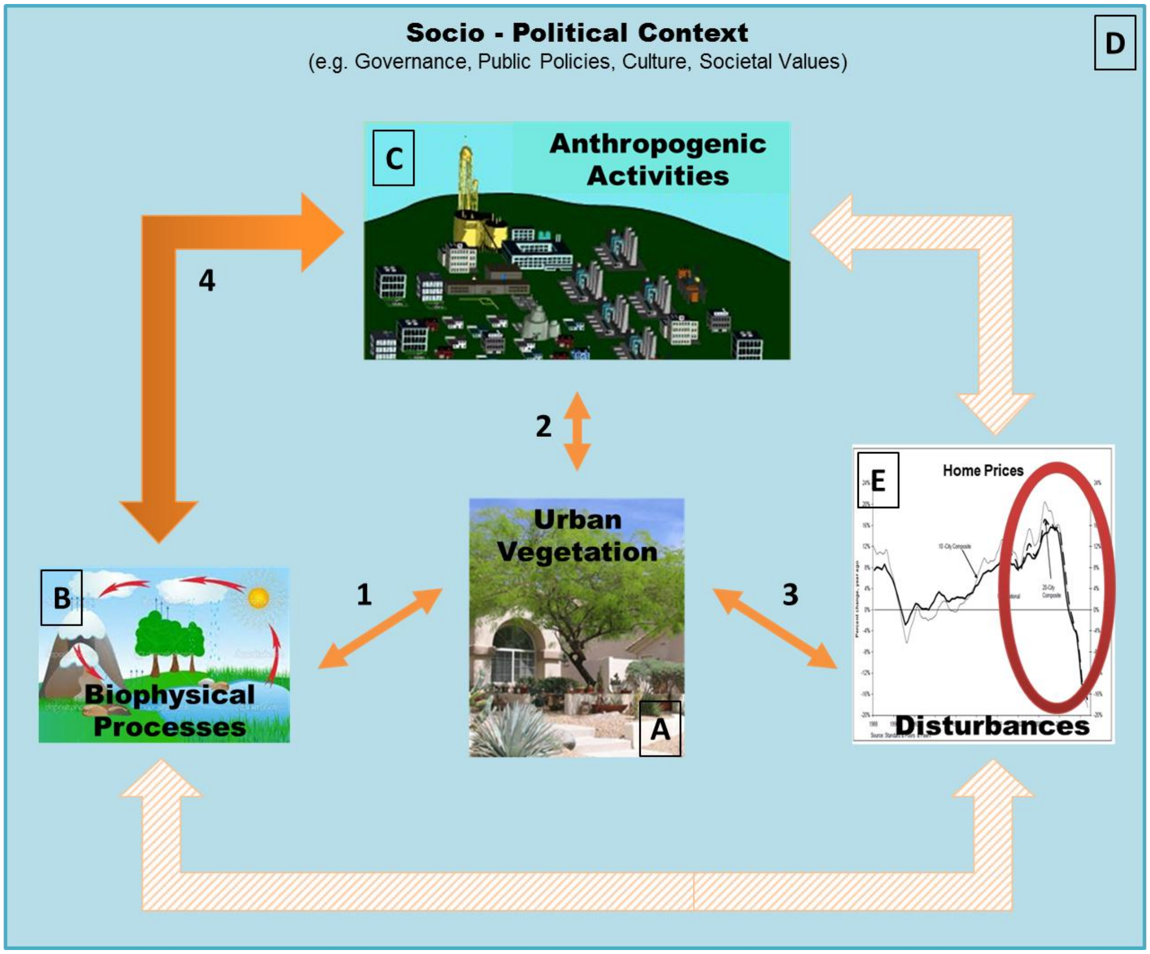

Figure 0.1 Conceptual framework for this dissertation research. Urban vegetation response to biophysical drivers (climate, substrate, biotic interactions), disturbance (biophysical and socioeconomic) and human management, where anthropogenic activities also respond to biophysical and socioeconomic factors.

CHAPTER 1

WHEN THE ECONOMIC ENGINE STALLS – A MULTI-SCALE COMPARISON OF VEGETATION DYNAMICS IN PRE- AND POST-RECESSION PHOENIX, ARIZONA, USA

ABSTRACT

Often, we think of cities as designed landscapes, where people manage everything from water to weeds. But we don't fully understand what happens to these extensively managed landscapes when there's an abrupt disruption in economic activity. Considering the ability of cities to support plant biodiversity and their importance as human habitat, we studied pre- and post-recession landscapes across a gradient of human influence by asking: How did vegetation change over time from before the housing bubble to after the nadir of the Great Recession? And how did vegetation vary across sites at regional versus residential scales? This investigation used long-term vegetation data to examine diversity trends and responses to a novel economic disturbance in an urban social-ecological system. Overall, we found that plant species diversity increased through time across scales, while species composition homogenized in urban and agricultural areas. Residential yards, however, initially had high compositional heterogeneity which then increased over time. Changes in residential diversity were driven by substantial increases in the role of annual plants. This research improves our understanding of spatiotemporal vegetation dynamics in a coupled human-natural system, and specifically how urban

vegetation dynamics are linked to anthropogenic influence. Ultimately, we recommend that city planners and managers consider economic trends when approaching community projects because of the interconnectedness of ecology and socioeconomics in urban landscapes.

1 Introduction

Vegetation has long been recognized as a primary contributor to ecosystem functioning and stability (Cardinale, Matulich, Hooper, Byrnes, Duffy, Gamfeldt, Balvanera, O'Connor, & Gonzalez, 2011; de Mazancourt, Isbell, Larocque, Berendse, De Luca, Grace, Haegeman, Polley, Roscher, Schmid, Tilman, van Ruijven, Weigelt, Wilsey, & Loreau, 2013; Grime, 1998) as well as to human health and well-being (Bolund & Hunhammar, 1999; Harlan, Brazel, Prashad, Stefanov, & Larsen, 2006). While the majority of studies examining vegetation composition and change have been in natural systems, a growing number are now aimed at understanding these same phenomena in social-ecological systems (SESs) (Johnson, Tauzer, & Swan, 2015; Kremer, Hamstead, & McPhearson, 2013) including urban ecosystems. Studies show linkages between vegetation diversity and socioeconomics in urban areas, and the phrase “luxury effect” was coined to capture the idea that higher socioeconomic status corresponds to the observed higher plant diversity in wealthier residential areas (Hope, Gries, Zhu, Fagan, Redman, Grimm, Nelson, Martin, & Kinzig, 2003). In Phoenix, Arizona, USA, neighborhood socioeconomic status predictsperennial diversity (Martin, Warren, & Kinzig, 2004) and spatially structures plant and bird diversity (Kinzig, Warren, Martin, Hope, & Katti, 2005).

In natural systems, drought or insect outbreaks are examples of exogenous disturbances that structure plant communities (Pickett & White, 1985). Non-urban SESs including forests, rangelands, and fisheries are increasingly stressed by global changes (Foley, DeFries, Asner, Barford, Bonan, Carpenter, Chapin, Coe, Daily, Gibbs, Helkowski, Holloway, Howard, Kucharik, Monfreda, Patz, Prentice, Ramankutty, & Snyder, 2005; Vitousek, Mooney, Lubchenco, & Melillo, 1997). Urban SESs are not exempt from comparable disturbances. In urban SESs, like the Phoenix metropolitan area (hereafter Metro Phoenix), natural disturbances are often mitigated by human actions such as supplemental watering and the use of pesticides. This results in highly managed and relatively stable urban landscapes (Knapp, Dinsmore, Fissore, Hobbie, Jakobsdottir, Kattge, King, Klotz, McFadden, & Cavender-Bares, 2012). Management itself, however, can be considered to be a form of disturbance (Ripplinger, Franklin, & Edwards, 2015), and management decisions are often influenced by socioeconomic drivers. For example, the recent housing recession that began in 2006 – sometimes termed The Great Recession – impacted the burgeoning Metro Phoenix area leading to high rates of foreclosure and unemployment. Viewed through the lens of the press-pulse disturbance (PPD) framework (Collins, Carpenter, Swinton, Orenstein, Childers, Gragson, Grimm, Grove, Harlan, Kaye, Knapp, Kofinas, Magnuson, McDowell, Melack, Ogden, Robertson, Smith, & Whitmer, 2011), management and design activities in cities are press-disturbances that provide sustained controls on diversity and composition of urban plant communities, while a sudden, unexpected shock like the Great Recession acts as a socioeconomic pulse-disturbance that can potentially affect urban vegetation from local to regional scales.

Urban ecological homogenization occurs when urban landscapes, even those designed to have different functions or meet different landscaping aesthetics, look more like each other than they do the surrounding natural landscapes (McKinney, 2006; Trentanovi, von der Lippe, Sitzia, Ziechmann, Kowarik, & Cierjacks, 2013). Under urban ecological homogenization (Knapp, Dinsmore, Fissore, Hobbie, Jakobsdottir, Kattge, King, Klotz, McFadden, & Cavender-Bares, 2012), natural vegetation assemblages are replaced by vegetation assemblages and urban ground cover like lawns or other types of gravel/bark cover (Cadenasso, Pickett, & Schwarz, 2007; Walker, Grimm, Briggs, Gries, & Dugan, 2009). Biotic homogenization has been attributed to transported landscapes (Anderson, 1952), the commercial nursery trade, real estate developer decisions, homeowner values and desires, and government and non-government regulations. National home improvement retailers contribute to homogenization by making available a globally derived, standardized stock of nursery plants, selected for gardening hardiness zones and homeowner appeal. Introduced ornamental species increase the occurrence of non-native and invasive species in cities (Reichard & White, 2001) and in-turn affect urban biodiversity and vegetation homogeneity. By designing ‘dreamscapes’ and establishing the initial conditions for housing developments (Larsen & Harlan, 2006), developers initially determine the underlying style and structure of residential landscapes. Homeowner’s associations (HOAs), increasingly common in rapidly developing areas, often have landscaping requirements that may be legally enforced, sometimes resulting in fines and in extreme cases even foreclosure (Lerman, Turner, & Bang, 2012; McKenzie, 1994). HOA regulations affect both landscaping form/function as well as the plant species used. These homogenizing processes are driven by human decisions, including top-down HOA regulations and bottom-up homeowner decisions.

The Great Recession dealt a particularly severe blow to the Metro Phoenix housing market, leading to a contraction in development, increased employment, and ultimately, high rates of foreclosures. In 2010 alone, there were 2.9 million foreclosure filings nationwide, roughly 6% of which were Arizona households (analytics from realtytrac.com). At the regional scale, we hypothesized the greatest recession impacts would be n the urban environment (versus agricultural or desert areas), in large part due to the recession's effect on landholders. Of the different types of urban land use, households are among the most intensively managed landscapes, and linkages between socioeconomic status and urban vegetation are well supported (Grove, Troy, O'Neil-Dunne, Burch, Cadenasso, & Pickett, 2006; Hope, Gries, Zhu, Fagan, Redman, Grimm, Nelson, Martin, & Kinzig, 2003; Larson, Casagrande, Harlan, & Yabiku, 2009; Luck, Smallbone, & O'Brien, 2009). At the residential level, we hypothesized that the Great Recession acted as a pulse of socioeconomic disturbance, leading to increased plant biodiversity in residential areas and increased biotic homogenization in urban areas, both due to spontaneously introduced annuals ("weeds") resulting from neglect of properties.

By using a core dataset from the Central Arizona-Phoenix Long-Term Ecological Research project (CAP LTER), we examined and compared how designed urban landscapes and their surrounding desert and agricultural counterparts varied in diversity and community composition, recognizing that these measures of community diversity are influenced by anthropogenic and biophysical factors to different degrees and at different scales. Specifically, we addressed the following questions: How did vegetation change over time from before the housing bubble to after the nadir of the Great Recession? And how did vegetation vary across sites at regional versus residential scales?

2 Methods

2.1 Study Area and Data

Metro Phoenix is home to nearly five million people and until 2008 was among the fastest growing metropolitan areas in the U.S. (US Census Bureau, 2000). The CAP LTER research program focuses on a region of 6400 km^2 in central Arizona that encompasses the entire Phoenix metropolitan area, as well as adjacent agricultural land and native Sonoran Desert landscapes (Grimm, Grove, Pickett, & Redman, 2000).

As part of the CAP LTER program, a regional survey of plant communities is regularly conducted during late spring at 204 sites (Grimm, Hope, Gries, & Martin, 2010). These “Survey 200” plot locations were selected using stratified random sampling in order to capture a gradient of human influence across Urban, Agriculture, and Desert land uses across the Metro Phoenix area (Fig. 1.1). To do so, one 30 x 30-m plot was randomly placed within each 4 x 4-km tessellation-grid square in the urban area and one survey point within one of three 4 x 4-km tessellation-grid squares in the desert surrounding the urban area. Vegetation species composition and abundance data were collected for each 30 x 30-m plot during each survey (see Hope et al., 2003 for full description). We analyzed abundance data for herbaceous, succulent, and woody plants based on surveys collected in 2000 and 2010 (bracketing the Great Recession).

Urban landscapes are predominantly planned landscapes, but design and management happen at different levels. Top-down controls include zoning, city planning, and homeowner association covenants, conditions, and restrictions. Bottom-up drivers include individual landscaping decisions, and socioeconomic status (Walker, Grimm, Briggs, Gries, & Dugan, 2009). There is no single appropriate scale for this type of analysis, so we focused our analyses at two spatial scales: (1) at the regional level where policy-makers manage for resources (ecosystem services), and (2) at the residential level where landscaping is managed on individual properties. At the regional level urban planning and design interacts with biophysical processes and urban governance. At the residential level householders make decisions about landscaping choices based on personal preferences and socioeconomics.

Broad regional land use categories were the basis of stratified sampling and were used to describe each Survey 200 site (i.e. Desert, Agricultural, Urban)(Hope, Gries, Zhu, Fagan, Redman, Grimm, Nelson, Martin, & Kinzig, 2003). Then, finer-scale within-urban residential categories were designated by CAP LTER scientists to describe household-level landscaping (i.e. Mesic, Oasis, Xeric) within a desert city. At the regional level, Desert sites included intact Sonoran Desert as well as mountain park preserves. Agricultural plots consisted of both subsistence and cash crops, including cotton, alfalfa, citrus, and cattle. Urban sites included the following land uses: commercial/industrial, transportation, city parks, and residential. At the residential level, Mesic ground cover primarily consists of lawn and lacks gravel swaths. Mesic yards often have broadleaf trees and shrubs and are characterized by low water use efficiency. Xeric plots are usually covered in gravel or another form of mulch and described as lacking lawns (turf grass). Xeric yards are often drip-irrigated and have desert-adapted, low water-use vegetation. Oasis plots are a mixture of the Mesic and Xeric landscaping motifs (Martin, Peterson, & Stabler, 2003).

2.2 Data Analysis

2.2.1 Richness, Heterogeneity

In order to compare diversity in 2000 and 2010 at both regional and residential land use levels, we computed plant species richness of each survey site and time period, and averaged by land use. We computed a Bray-Curtis distance matrix of all pairs of sites for both survey years, which was then used in the analysis of heterogeneity and in multivariate analysis of site composition. Following Collins (1992), we conducted heterogeneity analysis of plant communities over all possible pairs of sites, then for sites within the same land use type and survey year. We used species importance to determine which species dominated different land use types. Importance Values (IVs) were defined in this study as average abundance of each species in each land use type (Aho, Roberts, & Weaver, 2008). Analyses were done using the open-source software R supplemented by the ‘vegan’ and ‘labdsv’ packages (R Development Core Team 2011).

2.2.2 Non-metric Multidimensional Scaling

We determined how community composition varied among study sites for the two survey periods by using non-metric multidimensional scaling (NMDS). NMDS, an unconstrained ordination method, is a multivariate analysis based on species abundance data for survey sites, indirectly reflecting environmental processes that structure vegetation communities (Causton, 1988). The abundance variable from the Survey 200 data used in NMDS was plant species frequency and was not transformed. We used the Bray-Curtis dissimilarity metric and conducted 999 permutations. Analyses were performed with default options for the metaMDS function in the ‘vegan’ package in R open-source software (Oksanen, Blanchet, Kindt, Legendre, Minchin, O'Hara, Simpson, Solymos, Stevens, & Wagner, 2011). Permutational multivariate analysis of variance (PERMANOVA) calculates the significance of the resulting ordination models (Anderson, 2001). We used the Adonis function in the r package ‘vegan’ for PERMANOVA calculations of overall model significance, with Bray-Curtis as the dissimilarity measure (Clarke & Gorley, 2006; Magurran, 1988).

2.2.3 Proportion of Annuals

Short-lived annual species are adapted to respond to resource pulses, such as increases in space and decreases in competitors. Annual plants are not often used for landscaping in arid Phoenix and therefore annuals in the urban area represent opportunistic “weedy” pioneers that might typically be managed against in a well-maintained yard. To determine the dominance of short-lived, high-turnover plant species, we calculated the number of annual species on each land use type relative to the total number of species per land use type for each survey date. Percent annual species was calculated for all land use types at both regional and residential levels.

Changes in each measure of diversity and composition were examined across survey years at each land use stratum. Though the effects of the recession extended beyond 2010, we examined 2010 vegetation spatial patterns in greater detail for early post-recession changes.

3 Results

3.1 Regional Dynamics from 2000 to 2010

At the coarsest level of analysis – regional – survey sites showed an effect of time from 2000 to 2010. Mean plant species richness increased from pre- to post-recession for Agricultural ($\Delta Richness \cong 5$), Desert ($\Delta Richness \cong 18$), and Urban sites ($\Delta Richness \cong 12$) (Fig. 1.2a). Also, community heterogeneity was significantly different pre- and post-recession (Fig. 1.3a). Decreases in Urban and Agricultural community heterogeneity are particularly notable, given the increase in heterogeneity and richness observed at Sonoran Desert sites. The decrease in compositional heterogeneity (in other words, the increase in homogeneity) on anthropogenic land covers after the recession resulted from the pronounced increase in the percentage of annual plant species in the regional land covers following recession (Table 1), increases ranging from 18.7% in Urban to 46.5% on Agricultural sites.

Regional NMDS results (stress = 0.2512) revealed that Desert sites were composed of a consistent set of species across years and were largely distinct in their community composition from Agricultural and Urban sites, as indicated by the clustering of Desert sites from both years on axis 1 (Fig. 1.4a). Agricultural sites were compositionally similar to (scattered among) Urban sites across years (Fig. 1.4a), although they had less variation in composition than Urban sites. Compositional shifts along axis 2 were also evident between 2000 and 2010 for all three regional land use types (Fig. 1.4a). The differences among regional-level land uses and between years were significant, with ~12% of sums of squared differences explained by regional land use (PERMANOVA model with Bray-Curtis dissimilarity: r^2 = 0.129; p < 0.001) and ~7% sums of squared differences explained by survey year (r^2 = 0.072; p < 0.001).

3.2 Regional diversity and composition 2010

In the post-recession surveys (2010), plant species richness differed significantly among the three regional land use types (Fig. 1.2a). Species richness was highest on the Desert sites (N_{Desert} 28 ± 1, $T = -10.64$, $P = <0.01$ per 900 m^2), and lower on both anthropogenic site types (N_{Agri} 10 ± 1, $T = -1.14$, $P = <0.26$; N_{Urban} 21 ± 1 $T = -8.01$, $P = <0.01$per 900 m^2). In contrast, heterogeneity on Agricultural and Urban sites was similar (Het_{Agri} 0.92 ± 0.003, $T = -3.33$, $P = <0.01$; Het_{Urban} 0.91 ± 0.001, $T = -6.37$, $P = <0.01$ per 900 m^2) but showed significantly higher heterogeneity than Desert sites (Het_{Desert} 0.89 ± 0.002, $T = 3.29$, $P = <0.01$ per 900 m^2) (Fig. 1.3a). Heterogeneity was lower on Desert sites despite this land cover having the highest species richness, indicating a high degree of evenness among Desert sites that was lacking in Urban and Agricultural areas.

Different key species were associated with each of the three regional land uses in 2010 (see Appendix S1). The species of highest importance (IV ≥ 6) for Desert sites were all native perennials, including one tree species (*Parkinsonia microphylla*), and two shrub species (*Condea emoryi*, *Ephedra spp.*). Rather than the crop species themselves, the most important species on Agricultural sites was *Cupressus sempervirens*, an introduced ornamental tree commonly used in windblocks or as hedgerows between fields. Few cultivated species had consistently high abundance across Agricultural sites because there are many different cash and subsistence crops grown in and around Phoenix and because of the limited extent of this class in the sampling scheme. The one exception was *Citrus* spp., due to the widespread planting of citrus trees in agricultural settings as well as urban landscapes. The two most important species in Urban were trees with high horticultural value (*Parkinsonia aculeata*, *Beaucarnea recurvata*).

3.3 Residential Dynamics from 2000 to 2010

Mean plant species richness increased from pre- to post-recession for all three residential site types, but most dramatically for Mesic ($\Delta Richness \cong 14$) sites, as predicted (Fig. 1.2b). Richness increased to a lesser extent on Xeric ($\Delta Richness \cong 11$) and Oasis ($\Delta Richness \cong 9$) sites as well. Mesic sites experienced the greatest overall increase in richness, but Xeric sites had the greatest increase in proportion of annual species ($\Delta Richness_{\%} \cong 26.3\%$) (Table 1). Percent annuals increased less on Mesic ($\Delta Richness_{\%} \cong$

22.1%) and Oasis ($\Delta Richness_{\%} \cong 14.6\%$) sites. Heterogeneity increased on all residential land use types from pre- to post-recession (Fig. 1.3b). Similar species occurred in the plant communities at all three residential types (see Fig. 1.4b confidence ellipses) but were slightly different between the survey years. Residential NMDS results (stress = 0.2869) highlighted differences in composition across years but not among residential land use types (PERMANOVA model with Bray-Curtis dissimilarity: r^2 = 0.085; p < 0.001) (Fig. 1.4b). Within-year compositional similarities existed among Mesic, Oasis, and Xeric sites for both 2000 and 2010 surveys. Mesic sites were composed of a more similar suite of species, as indicated by the location of Mesic sites from both years below zero on axis 2 (Fig. 1.4b).

3.4 Residential Diversity and Composition 2010

Post-recession species richness at the residential level was higher on average than at the regional level. Lowest richness in 2010 was on the Mesic sites (N_{Mesic} 22 ± 3, T = 5.08, P = <0.01) (Fig. 1.2b). Highest richness was found on the Xeric sites (N_{xeric} 25 ± 3, T = 2.98, P = <0.01), and richness on the hybrid landscapes of Oasis sites was N_{Oasis} 24 ± 2 (T = 2.63, P = <0.01), but these differences were relatively small and not statistically significant. However, Oasis sites were significantly more heterogeneous (Het_{Oasis} 17 ± 0.15, T = 4.10, P = <0.01) than Mesic and Xeric site heterogeneity (Het_{Mesic} 15 ± 0.35, T = 4.94, P = <0.01; Het_{Xeric} 16 ± 0.30, T = 3.03, P = <0.01) in 2010 (Fig.1.3a).

Species importance values (IVs) quantified differences in the key species for the three residential land use site types (see Appendix S1). Species of highest importance (IV

≥ 6) on Mesic sites consisted of both native and introduced species. They included three ornamental tree species (*Acacia farnesiana*, *Beaucarnea recurvata*, and *Ebenopsis ebano*), a horticultural shrub species (*Gardenia jasminoides*), and one succulent genus (*Agave spp.*). The most important species on Oasis sites included the popular golden barrel cactus (*Echinocactus grusonii*) introduced from Mexico, the deciduous fruit tree (*Prunus persica*), and two introduced tree species (*Callistemon viminalis*, *Phoenix roebelenii*). Primarily native species were important on Xeric sites, as might be expected in a desert-like landscape, though none of them were cactus species but instead trees (*Parkinsonia aculeata*, *Parkinsonia florida*, and *Populus fremontii*). Also important at Xeric sites was the introduced tree genus *Eucalyptus*.

4 Discussion

In this observational study, we compared the diversity and composition of plant communities before and after the Great Recession, investigating multi-scale vegetation patterns along a gradient of human influence. Consistent with our predictions, we found changes from pre- to post-recession in urban plant communities, specifically diversity increased across scales for all land use types while vegetation homogenization occurred for urban and agricultural sites. At the regional scale, Desert locations experienced the steepest increase in total species richness over the study period (Fig 1.2a), whereas Agricultural and Urban locations exhibited steep increases in vegetation homogeneity (Fig 1.3a). At the residential level, annual species surged in importance on all residential site types (Table 1) and weedy plants ranked highly among the important species

(Appendix S1), especially in Mesic yards where nearly half of the important species were weedy annuals. Our results suggest underlying differences in drivers of directional change between 2000 and 2010, with regional environmental drivers (i.e. precipitation) most likely responsible for decadal change in Desert plant communities, and the pulse of socioeconomic disturbance associated with the Great Recession most likely prompting change in Urban and Agricultural areas.

The luxury effect hypothesis predicts higher diversity will correspond to higher income (Hope, Gries, Casagrande, Redman, Grimm, & Martin, 2006; Hope, Gries, Zhu, Fagan, Redman, Grimm, Nelson, Martin, & Kinzig, 2003). However following decreases in income and home values with the Great Recession, we found increased plant species richness and distinct community composition in Urban and Agricultural locations despite the fact that the housing market hit record lows and the 2010 survey followed four years of recession. Post-recession diversity differences could have been caused by background increases in Desert plant species richness from 2000 to 2010. However, annual plant species increased disproportionately at Urban and Agricultural sites following the Great Recession (Table 1). The increase in percent annuals and species richness at residential survey sites post-recession were not congruent with the luxury effect hypothesis.

Urban landscapes consist of numerous land uses and large species pools. And yet, our heterogeneity analysis revealed significant increases in ecological homogeneity of Urban and Agricultural locations following the Great Recession (Fig 3a). When urban landscapes - even those designed to perform different functions or satisfy different aesthetics - are more like each other than they are the surrounding natural landscapes, ecological homogenization has occurred (McKinney, 2006; Trentanovi, von der Lippe, Sitzia, Ziechmann, Kowarik, & Cierjacks, 2013). At a regional scale, our results reflected increases in homogeneity of anthropogenic land uses (Fig 3a). At the residential scale, yards of the three different land uses appeared physically quite different from one another. Xeric yard designs consisted of sparse shrubs and rocky ground cover, while Mesic yards had grassy expanses and verdant trees. Rather than being designed for native or non-native species, the three residential land uses were more likely chosen for ease of maintenance or to control for water use. So despite the contrasting physical appearances of these landscaping aesthetics, we found the plant community composition of the three residential site types was similar (Fig 1.4b), with overlapping composition between survey years, even while the percentage of annual species increased (Table 1.1).

The unexpected compositional similarity we uncovered between Xeric and other residential landscape types has not been found in previous studies (Hope, Gries, Casagrande, Redman, Grimm, & Martin, 2006; Hope, Gries, Zhu, Fagan, Redman, Grimm, Nelson, Martin, & Kinzig, 2003; Martin, Warren, & Kinzig, 2004; Walker, Grimm, Briggs, Gries, & Dugan, 2009). We expected water-wise Xeric yards were designed to mimic the arid Sonoran Desert, and so would be distinct from other residential landscapes and more similar to Desert sites. Comparing across scales of analysis, Xeric yards had a different suite of key species than the surrounding native Sonoran Desert (Appendix S1) (only *Parkinsonia* spp. and the invasive *Cenchrus setaceus* in common among abundant species (IV >= 4)). Similar studies have recently shown how during urban ecological homogenization (Knapp, Dinsmore, Fissore, Hobbie, Jakobsdottir, Kattge, King, Klotz, McFadden, & Cavender-Bares, 2012), natural vegetation assemblages were replaced by popular vegetation assemblages and lawns or other types of gravel/bark ground cover (Cadenasso, Pickett, & Schwarz, 2007; Walker, Grimm, Briggs, Gries, & Dugan, 2009). While other studies have focused on comparisons among cities, we observed the homogenizing influence of recession within a single city, across regional anthropogenic land uses and following an economic disturbance. Our results add momentum to the call for a better understanding of the processes driving urban ecological homogenization over time and at multiple scales.

In conclusion, our data suggest that Phoenix Metro vegetation composition and diversity changed from 2000 to 2010. These changes were most likely initiated by the either the ‘press’ of normal biophysical variability (e.g. seasonal precipitation) or by the ‘pulse’ of an exogenous disturbance (e.g. the Great Recession). But it is unlikely that antecedent precipitation (57 mm for 2000 and 55 mm for 2010) contributed to increased diversity because it was slightly below average (61 mm) in both survey years. The pulse disturbance of the Great Recession would permit increased richness through lack of land management and increased homogeneity due to the increased role of annual plant species. The results presented here identify opportunities for additional research aimed at empirically identifying drivers of changes in urban plant communities and further studying press-pulse disturbances relevant to urban ecosystems. We recommend that urban planners and managers be cognizant of socioeconomic trends, like the Great Recession, that may generate pulse disturbances, since socioeconomics may result in ecological perturbations such as those observed in this study.

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Table 1.1. Percent of annual species relative to total species richness in 2000 and 2010. Results given for each land use type at both regional and residential levels.

2000							
REGIONAL	**Annuals (N)**	**Total (N)**	**Percent**	**RESIDENTIAL**	**Annuals (N)**	**Total (N)**	**Percent**
Agriculture	13	29	44.8	Mesic	23	126	18.3
Desert	56	124	45.2	Oasis	18	104	17.3
Urban	56	233	24.0	Xeric	11	96	11.5
2010							
Agriculture	63	69	91.3	Mesic	76	188	40.4
Desert	173	257	67.3	Oasis	80	251	31.9
Urban	198	464	42.7	Xeric	68	180	37.8

Figure 1.1. Map representing the spatial distribution of Survey 200 sites in Phoenix Metro and surrounding area (*N* = 204). Colored points indicate residential sites surveyed in 2010 by landscaping type.

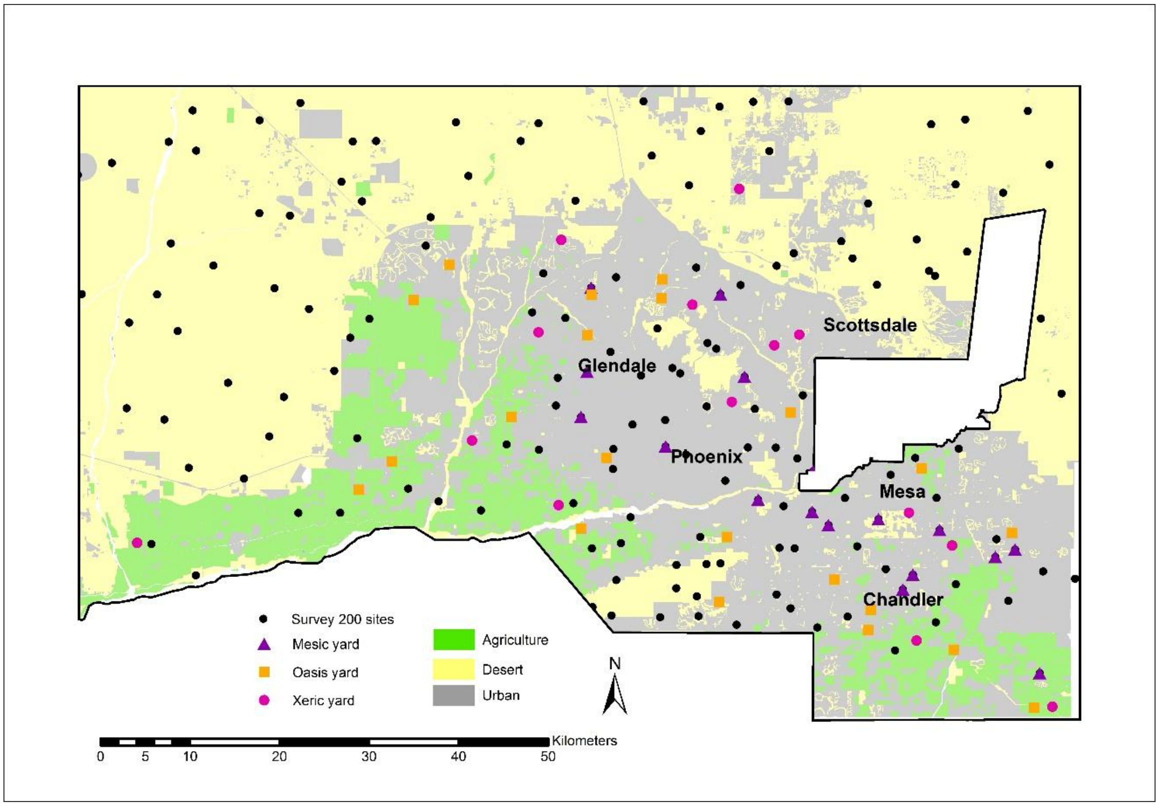
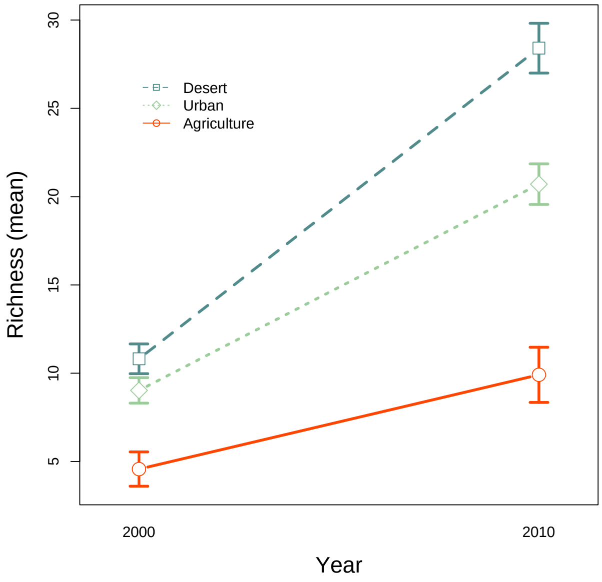
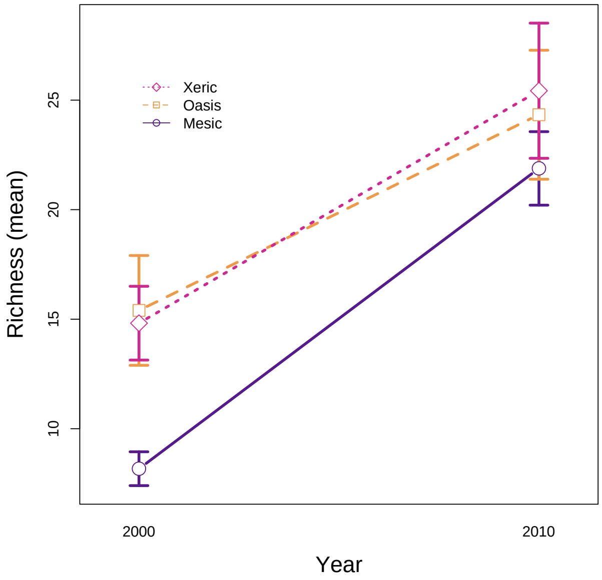


Figure 1.2. Mean (plus standard error) plant species richness of Survey 200 sites in 2000 and 2010 by (a) regional and (b) residential land use type.

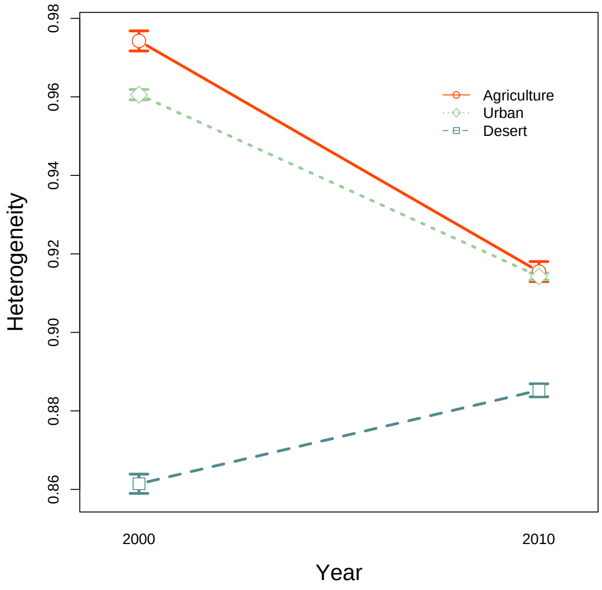


(a) REGIONAL

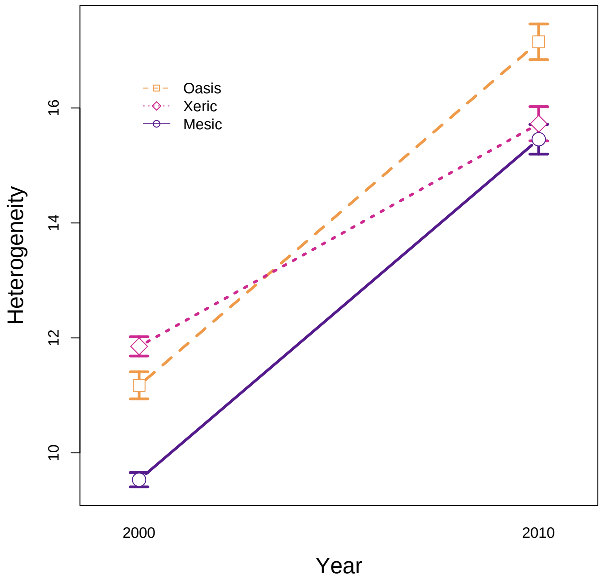


(b) RESIDENTIAL

Figure 1.3. Mean (plus standard error) vegetation heterogeneity of Survey 200 sites in 2000 and 2010 by (a) regional and (a) residential land use type. Heterogeneity analysis on all possible pairs of plots. Higher values of heterogeneity = higher spatial heterogeneity.

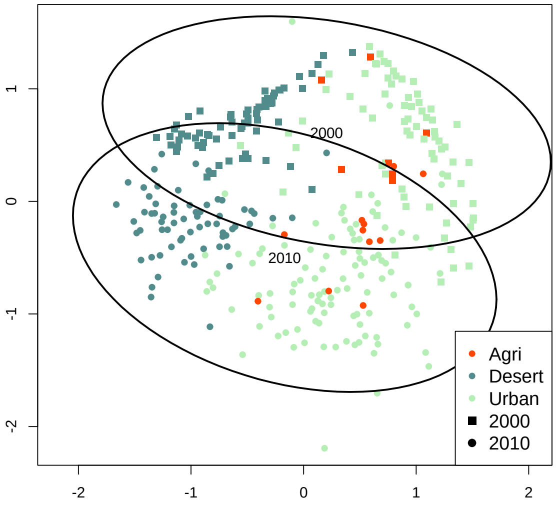


(a) REGIONAL

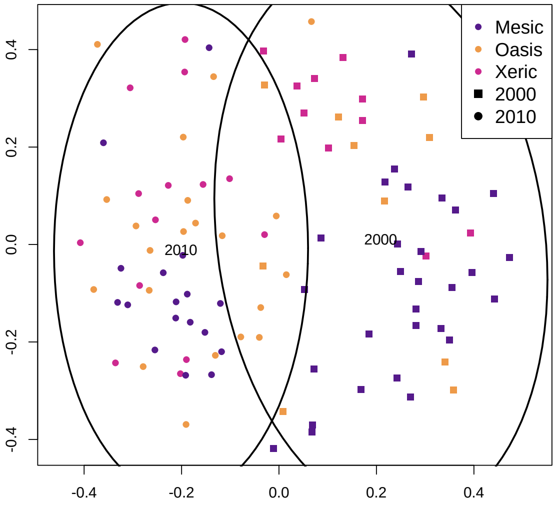


(b) RESIDENTIAL

Figure 1.4. Non-metric multidimensional scaling (NMDS) plots of Survey 200 sites by (a) regional and (b) residential land use type, using plant species abundance and Bray-Curtis dissimilarity. Symbols represent site location; symbol colors represent land use type and symbol shape represents year surveyed.



(a) REGIONAL



(b) RESIDENTIAL

CHAPTER 2

BOOM-BUST ECONOMICS AND THE ECOLOGY OF CITIES: HOW STRONG IS THE LINK?

ABSTRACT

In cities, human activities like supplemental watering and plantings of ornamental species are thought to decouple vegetation diversity from biophysical processes. Consequently, socioeconomics are arguably the most important factor governing vegetation in urban ecosystems. Socioeconomic disturbances, like The Great Recession of 2007-2010, disrupt normal social and economic activity causing changes to the ecology of cities that have yet to be examined. Using Phoenix, Arizona, USA as a case study, we explored the dynamics of residential vegetation diversity from before to after The Great Recession. Our findings linked plant composition and increasing richness with the housing boom of 2001-2006. We were surprised to find, however, that annual plant species did not respond to socioeconomic disturbance but instead were linked to winter precipitation similar to nearby native desert plant communities. Cross-site comparisons are necessary to further elucidate the extent to which our findings hold across older and newer, mesic and arid, and growing and shrinking cities.

INTRODUCTION

As the global human population becomes increasingly urban, urban flora and fauna provide vital opportunities for people to interact with nature. For some people the urban setting provides their only opportunity to connect with nature, and close to home is often where those interactions occur. Highly managed and designed urban landscapes like yards, gardens, and parks offer a cultivated form of nature that have been found to enhance human well-being and feelings of connection to nature (Mayer and Frantz 2004), and in turn, exposure to nature often promotes pro-environmental behavior (Geng et al. 2015). In other words, when people connect with what they consider to be pleasing forms of nature, they are likely to be physically (Maas et al. 2006) and mentally healthier (Fuller et al. 2007), and engage in sustainable behaviors leading to improved ecosystem health (Tzoulas et al. 2007). Like their non-urban counterparts, urban ecosystems are highly dynamic. Researchers are beginning to understand residential vegetation in a socioecological context (Larsen and Harlan 2006, Larson et al. 2009), and to apply understanding of biophysical disturbance and vegetation dynamics concepts (Pickett and White 1985) to urban ecosystems. Yet to date there has been little work on how socioeconomic drivers might affect urban vegetation dynamics.

Links between residential vegetation and socioeconomic factors are beginning to be well established (e.g. Martin et al. 2004, Kinzig et al. 2005, Luck et al. 2009). Socioecological concepts have been developed to describe these linkages, like the "ecology of prestige," which connects residential vegetation to the capacity of a household to manage vegetation. Variable capacity among particular social strata, for example, may result in variable preferences e.g. mature trees, established lawns, or perennial gardens (Grove et al. 2006, Grove et al. 2014). As a consequence, the desire of a given household to be associated with a particular social stratum is manifested in the very public display of their front-yard vegetation. Indeed, Hope et al. (2003) termed the positive relationship between household income and plant diversity the “luxury effect”. The basis of this correlation is that people choose to occupy (or install) higher diversity landscapes as their socioeconomic status increases. Such studies account for the underlying financial capacity of a household to regulate vegetation cover and diversity in their yards.

In a nutshell:

- Socioeconomic disturbance is a profound change to a social-ecological system that is caused by a relatively abrupt disruption to economic and social activity
- To successfully manage widespread urban vegetation, we need to understand the relative importance of not only biophysical but also socioeconomic drivers, with particular attention to boom-bust cycles in the economy
- We analyzed the drivers of residential vegetation richness and composition in Phoenix, AZ, before, during, and after the Great Recession
- The housing boom had a stronger effect on plant species richness than the bust
- Precipitation remains an important driver of annual plants

In urban ecosystems biophysical disturbances, caused for example by floods, fire, and pests, are well-studied, and are known to erode urban stream beds (Walsh et al. 2005), alter soil resources (Pickett and Cadenasso 2009), and increase human-wildlife conflict and destruction of homes by wildfires (Radeloff et al. 2005). For many types of disturbance, however, viable engineered solutions exist. For instance, cities in the US Southwest have had success in flood management through the installation of constructed retention basins (Grimm et al. 2005). But a potentially important driver of urban vegetation dynamics has been overlooked – socioeconomic disturbance. We differentiate this driver from anthropogenic disturbance, which is generally considered to be a detrimental physical impact on the environment (e.g., forest clear-cutting). Drawing on the concept of ecological disturbance (Pickett et al. 1989), we define socioeconomic disturbance as ***a profound change to a social-ecological system that is caused by a relatively abrupt disruption to economic and social activity***. Here we focused on ecological changes in an urban ecosystem, but we propose that disturbances can be conceptualized the same way whether the driver is socioeconomic or biophysical.

Pioneer plant species rely on disturbance to establish, have broad physiological tolerances, are able to acclimate to a wide range of conditions (e.g. weedy introduced species), and have rapid growth rates (e.g. annual plants) (Rejmanek and Richardson 1996). In urban systems, when people stop maintaining landscapes pioneer species are presented with an opportunity to increase in abundance both locally and spatially. Additionally, successful invaders often go through multiple introduction events (Gray 1986), and adjacent landscapes under economic stress may provide a pool of colonizing species. Colonization pressure from annual and introduced species would be expected to increase on landscapes with adjacent unmaintained landscapes.

We studied the effect of socioeconomic disturbance on the dynamics of plant species richness and composition in residential landscapes across the Phoenix metropolitan area, Arizona, USA. The Central Arizona-Phoenix Long-Term Ecological Research (CAP LTER) project conducted a comprehensive survey of urban vegetation composition in 2000, 2005, and 2010, providing a unique opportunity to examine the effects of extensive mortgage foreclosures and other impacts of the Great Recession on residential vegetation. In residential areas there is a direct connection between the 'land manager' (home owner or property manager) and the landscape, so we hypothesized that we would detect effects of the Great Recession on the residential landscape. Our objective was to examine the effect of socioeconomic disturbance factors versus biophysical variables on plant species diversity and composition. We expected socioeconomic disturbance to have a stronger influence than biophysical variables because urban vegetation is controlled by decisions by homeowners more than by climate or edaphic factors (Seto and Kaufmann 2003, Cook et al. 2012). We further tested whether plant species richness and composition differed from pre- to post-recession. We expected the Great Recession to indirectly release landscapes from direct human controls like herbicides, weeding, and horticultural plantings, such that annual and early successional species, in particular, would increase in abundance and distribution. We asked the following questions: Do socioeconomic disturbance factors (e.g. foreclosed home value, foreclosure density) predict residential plant diversity? How does the strength of these factors change with socioeconomic disturbance (recession) and compare to biophysical drivers? Are there differences in how changes in diversity relate to socioeconomic status through time? How does urban vegetation diversity change through a period of economic boom-and-bust?

METHODS

Study area and data

The Phoenix Metropolitan Area (hereafter ‘Phoenix’) consists of the city of Phoenix, AZ, USA along with several smaller adjacent cities, and is home to more than 4 million people. The CAP LTER study area is 6,400 km^2 and encompasses the Phoenix Metropolitan Area as well as surrounding agriculture land and the Sonoran Desert ecoregion. From the CAP LTER systematically random designed ‘Survey 200’ (Hope et al. 2003), we selected 119 plots that were classified as residential land-use from throughout the Phoenix area. Each plot consisted of a single 30 x 30 m area where vegetation is re-surveyed every five years (2000, 2005, 2010) and sits a minimum of 500 m from the nearest adjacent plot. Plots were classified as residential if surveyors assigned >50% of the plot to a residential type land-use, for example, single-family or multi-family residential. For each plot, all vascular plant species were identified and the number of individuals counted. Species were assigned to the following life form categories: (i.e. herbaceous annuals, cacti/succulent species, shrubs/hedges, and trees).

Data from the vegetation surveys were used to calculate the response variables -- measures of plant diversity and composition. We then identified socioeconomic and biophysical predictors hypothesized to affect residential vegetation in a desert city based on the literature and ecological theory. Variables were selected to assess the relative importance of dynamic socioeconomic disturbance and biophysical factors, and were chosen to correspond to the time periods of interest. We considered 19 predictors of the diversity and composition of plant species (see Table 1). Selected variables were hypothesized to be factors affecting urban and global patterns of vegetation diversity; e.g. neighborhood socioeconomic status (Hope et al. 2003, Kinzig et al. 2004, Martin et al. 2005), legacy effects of land-use (Cook et al. 2012, Johnson et al. 2015), as well as fundamental climate and biophysical factors (Grace 1999). Socioeconomic predictors specific to foreclosures resulting from the Great Recession were taken from publicly recorded real estate data compiled by The Information Market (www.theinformationmarket.com). Home values were taken for foreclosed homes within 500 m of each plot – a distance selected to prevent overlap among plots and to correspond to 'neighborhood' size used in other studies (York et al. 2011).

Statistical analysis

To examine the relative importance of socioeconomic and biophysical variables in driving residential vegetation diversity over time, we developed a series of generalized linear regression models where richness (for all plants, annuals only, and introduced species only) across all plots and survey periods was modeled as a function of several predictor variables (Table 1).

Analyses of change in plant community composition were based on abundance data for all plant species. Hierarchical cluster analysis, using Euclidean distance and Ward's method (Ward 1963), was used to identify distinct groups of plots based on species composition for each survey year separately. Ward's method of hierarchical agglomerative clustering progressively merges clusters to minimize within-group variance. Changes in the dominant species among clusters over time and cluster membership of plots between survey years were identified. The constrained ordination, Canonical Correspondence Analysis (CCA) (Ter Braak 1986), was used to relate variation in plant species composition to socioeconomic and biophysical variation in the environment, and also to examine relationships among groups identified by hierarchical cluster analysis.

To examine trends in diversity over the time period spanning the Great Recession, species richness in each survey year was calculated for the 119 plots in three ways – (1) for all plant species, (2) for annual species only, (3) for introduced species only. These subsets of annual and introduced species allowed us to explore our hypotheses about their abilities to respond to management release. Analyses that used species richness were also conducted using Shannon diversity index, but results were nearly identical so for simplicity only species richness results are reported.

RESULTS

Drivers of residential vegetation diversity

Once highly correlated variables were excluded, the following predictors were included in generalized linear models (GLMs): winter precipitation, survey year, time since foreclosure, year house was built, assessed property value at foreclosure, and nearby foreclosure density (Table 1). Time since foreclosure, year built, and foreclosure density were not significant in any of the models for any of the richness response variables.

Survey year and assessed value at foreclosure influenced total species richness positively, while winter precipitation had a negative effect on richness (Table 1). For annuals, survey year affected total richness positively and winter precipitation had a negative effect on total richness (Table 1). Survey year and assessed value at foreclosure positively influenced introduced species richness, while winter precipitation had a negative effect on introduced species richness (Table 1). All other independent variables were non-significant for all datasets.

Community Composition

Plant species composition was in fact strongly structured by survey year, assessed home value at foreclosure, winter precipitation, and density of nearby foreclosures (Fig. 1). Plots were arranged along the x-axis by the negatively correlated variables of increasing winter precipitation and decreasing foreclosure density, and along the y-axis by the negatively correlated variables of increasing value of nearby foreclosed homes and decreasing (recent → earlier) date of foreclosure. Hierarchical clusters of plots based on plant species composition were structured in ordination space. Years 2000 and 2010

Cluster 1 plots tend to have homes that were built earlier, lower home values, and lower density of nearby foreclosures, with earlier foreclosure dates than Cluster 2 plots of all years. Year 2005 plots across all clusters were more often associated with high-value foreclosed homes, high density of nearby foreclosures, high winter precipitation, and more recent foreclosure dates. Cluster 3 plots in all years tend to be associated with high-value foreclosed homes that foreclosed more recently across the spectrum of foreclosure density and housing age.

Vegetation change

The residential surveys combined across all years included 611 plant species. The total number of species found in each survey year (gamma diversity of the residential landscape) trended upwards from 2000 to 2005 to 2010 (Fig. 2, Appendix Table S1). Site-level annual species richness peaked in 2005, while total and introduced species richness increased each survey year (Fig. 2).

In 2000, the Citrus-Fan Palm association, the largest group of 27 plots, was dominated (defined as the ~2 species with the highest count of individuals) by *Washingtonia* spp., a widely planted ornamental palm, and species of introduced ornamental fruit trees (*Citrus* spp.) (Fig. 3). The Flower Garden association, the second-largest group of 8 plots, was dominated by common introduced flower garden species, like roses (*Rosa* sp.) and lavender (*Lavandula* spp.). Finally, a third cluster of outlier plots (N=4) was dominated by the native prickly pear cactus, *Opuntia* spp., and the introduced horticultural flower, *Crocus* sp.

In 2005, the Acacia-Plantago association formed the largest group of plots (N=31) that were dominated by the small, native horticultural tree, *Acacia farnesiana*, and the weedy introduced herbaceous forb *Plantago* sp. This cluster included plots from the Citrus-Fan Palm and Flower Garden clusters of 2000 (Fig. 3). Desert Ornamentals, the second largest group (N=10), was dominated by *Leucophyllum frutescens* and *Nerium oleander*, both introduced perennials with horticultural value and long-lived flowers. The weedy *Plantago* spp. and the ornamental introduced *Aloe* spp. were also common in this plant association. Plots in this group came primarily from Citrus-Fan Palm and Opuntia-Crocus clusters in 2000. Finally, a small group of outlier plots (N=4) was dominated by the native but often weedy *Isocoma acradenia*.

In 2010, most plots remained in their 2005 cluster membership, but the dominant species changed over time (Fig. 3). The largest grouping (N=25) was the Saltbush-Oleander association dominated by *Atriplex elegans*, a native but weedy shrub, and the introduced ornamental tree/shrub, *N. oleander*. The Desert Ornamentals group (N=6) was dominated by *Aloe vera* and *Lantana camara*, both popular introduced ornamental species, and also *Nolina microcarpa*, a native perennial related to agaves that is often used as an ornamental. Finally, the Natal Plum-Palo Verde cluster consisted of the same outlier plots (N=4) as in 2005, but dominants changed to the introduced ornamental shrub Natal Plum *Carissa macrocarpa*, and the native ornamental tree Palo Verde *Parkinsonia aculeata*.

To look more closely at how richness of annual plants and introduced species changed through time, we plotted the distribution of site-level species richness for clusters for each survey year (Fig. 4). Clusters 1 and 2 included the majority of survey plots each year, with Cluster 3 consisting of fewer than 5 outlier plots in each year. For annual species (Fig. 4a), all Clusters had peak richness in 2005, where change in Cluster 3 was relatively small compared to the majority of survey plots, and annual species richness in this small cluster was relatively stable across survey years. For introduced species (Fig. 4b), richness in clusters 1 and 2 again increased across survey years, while introduced species richness in Cluster 3, the small group of outliers, had extremely high variance and therefore no significant trend over time.

DISCUSSION

Our study aimed to investigate the impact of the Great Recession on urban residential plant communities. We found that a component of boom-bust economics – assessed value at foreclosure – was a strong driver of plant community richness (Table 1) and composition (Figs. 1, 3). We also found that composition and species richness changed over the course of the housing bubble (from pre-2000 to 2007) to the bust of the Great Recession (from 2007 to post-2010) (Figs. 1-4).

Our study demonstrates that socioeconomic disturbances can have significant effects on residential plant diversity. From 2000 to 2010, however, we found that the biggest vegetation change detected came with the housing bubble, rather than the bust. The boom-bust economic disturbance did affect richness of annual and introduced species, but the responses of these functional types differed (Fig. 4). Annual plant species richness increased with the housing boom (evident in differences from 2000 to 2005 surveys), while introduced species richness increased following the Great Recession (from 2005 to 2010 surveys; Fig.2). The decrease in annual plants detected following the Great Recession could result from the survey in 2010 occurring too soon after the housing bubble ended to reflect its full effect on residential vegetation (refer to Fig. S1). Future surveys may reveal the expected increase in pioneer species. Plant species composition of plots changed most with the housing boom, and site composition was influenced predominantly by the value of nearby homes at the time of foreclosure (see Fig. 1). It appears that the particular five-year intervals captured in Survey 200 were better able to detect the "luxury effect" of the housing bubble than the ecological release of pioneer species that we predicted to occur when the bubble burst. The high-end homes that demonstrate this 2000-2005 luxury effect were, however, not immune from the subsequent increase in foreclosure rates (unpublished data).

Assessed value of foreclosed homes was also a strong predictor of total and introduced plant species richness, but importantly, winter precipitation was an equally strong predictor (Table 1). Biophysical factors like precipitation remain an important driver of annual vegetation diversity in urban ecosystems, whereas many introduced species in arid cities are tropical in origin and rely heavily on supplemental watering to persist.

Deserts are water-limited systems, so during wet years, increases in precipitation typically lead to increases in productivity. High seasonal precipitation that occurred in the months prior to the 2005 field survey (Fig. S2) could promote an increase in vegetation productivity, which in some studies has been shown to result in increased vegetation diversity (Mittelbach et al. 2001). In our study, an increase in annual plant species richness corresponded to the high precipitation period surveyed in 2005. This increase in richness was evident across all site types, regardless of cluster designation (Fig. 4a). Annual plants are adapted to reproduce quickly giving them the ability to respond to short-term pulses in precipitation. Higher diversity of annual species in 2005 could also be due to the housing bubble (Fig. S1), leading to increased economic resource availability for landscape plantings of ornamental annuals. However, the diversity of introduced species did not increase over the same time period (Fig. 4b), nor did total plant species richness (Fig. 2). Instead, environmental and economic conditions resulted in richness gradually increasing from 2005 into 2010 for all species and for introduced species (Fig. 2), with the lowest total and introduced species richness occurring on plots classified as outliers in their species composition (Fig. 4b). This could be a result of yard preferences or homeowner's association rules and norms, though this explanation is less likely for household preferences (Larsen and Harlan 2006, Yabiku et al. 2008, Larson et al. 2009). Also, in a stable or booming human-managed system, increases in spontaneous vegetation due to high precipitation likely result in increases in management activities like weeding, causing lower than expected introduced species richness such as in 2005 during a high-precipitation period.

This paper includes a number of intriguing results that we hope will stimulate additional research. First, it is puzzling that the housing bubble rather than the Great Recession had a stronger effect on plant community diversity and composition. This surprise may be due to the fact that the 2010 survey occurred early in the post-bubble housing slump (e.g. foreclosure rate in 2010 was 39%, as contrasted with average foreclosure rate of ~5%)(https://asunow.asu.edu/content/final-phoenix-area-foreclosure-numbers-2011). A lag in response-time required for the vegetation to change could account for a smaller-than-expected recession signal in the 2010 survey (Essl et al. 2015). Future surveys might reveal new patterns of vegetation change following the boom-bust. Second, we found that 2005 plant community composition was most similar to vegetation found at outlier plots. One possible explanation for this finding is that personal preferences tend toward yards with higher native species diversity, but in 2000 and 2010 without boom-time economics in play, average households could not afford to plant more native species. Future research into landscaping preferences among various demographic groups would further clarify this finding (e.g. Uren et al. 2015). Finally, previous research has shown that managed vegetation can be decoupled from the influence of precipitation (Buyantuyev and Wu 2012), but we found that antecedent precipitation played a much larger role in predicting species richness than expected. In particular, we noted that annual plant species did not respond to socioeconomic factors but instead they were highly correlated with winter precipitation much like native desert annual communities.

Here we have shown that although people heavily manage and design urban landscapes, a socioeconomic disturbance – like the recent housing boom-bust of the Great Recession – can be a strong driver of changes in plant community richness and composition. Additionally, in an arid city, biophysical factors remain an important driver of annual species abundances regardless of human activities. Undertakings aiming to maintain or increase urban biodiversity for its associated ecosystem services and improved human well-being need to systematically approach the effects of socioeconomic fluctuations on urban flora. Cross-site comparisons will be key to developing a broader understanding of these coupled dynamics across older and newer, mesic and arid, and growing and shrinking cities.

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	Predictors	**Responses**		
		Total Richness	Annuals Richness	Introduced Richness
Biophysical	Aspect			
	Elevation			
	History of agricultural land-use			
	Precipitation (summer)			
	Precipitation (winter)	303.45***	76.64***	204.86***
	Slope			
	Soil type			
	Temperature (maximum)			
	Temperature (minimum)			
Temporal	**Survey year**	101.36***	150.90***	40.30***
	Original purchase date			
	Time since foreclosure			
	Time since land-use change			
	Year built			
Socioeconomic	**Assessed value at foreclosure**	296.03***		222.61*
	Foreclosure density			
	Original mortgage value			
	Original purchase price			
	Property size			

Table 2.1. Complete list of possible predictor variables prior to exclusions due to multicollinearity. Variables included in final GLM analysis denoted by **bold** text. Level of significance denoted by asterisks (*** ≤ 0.001, ** ≤ 0.01, * ≤ 0.05). Test statistic X^2 reported only for significant predictors.

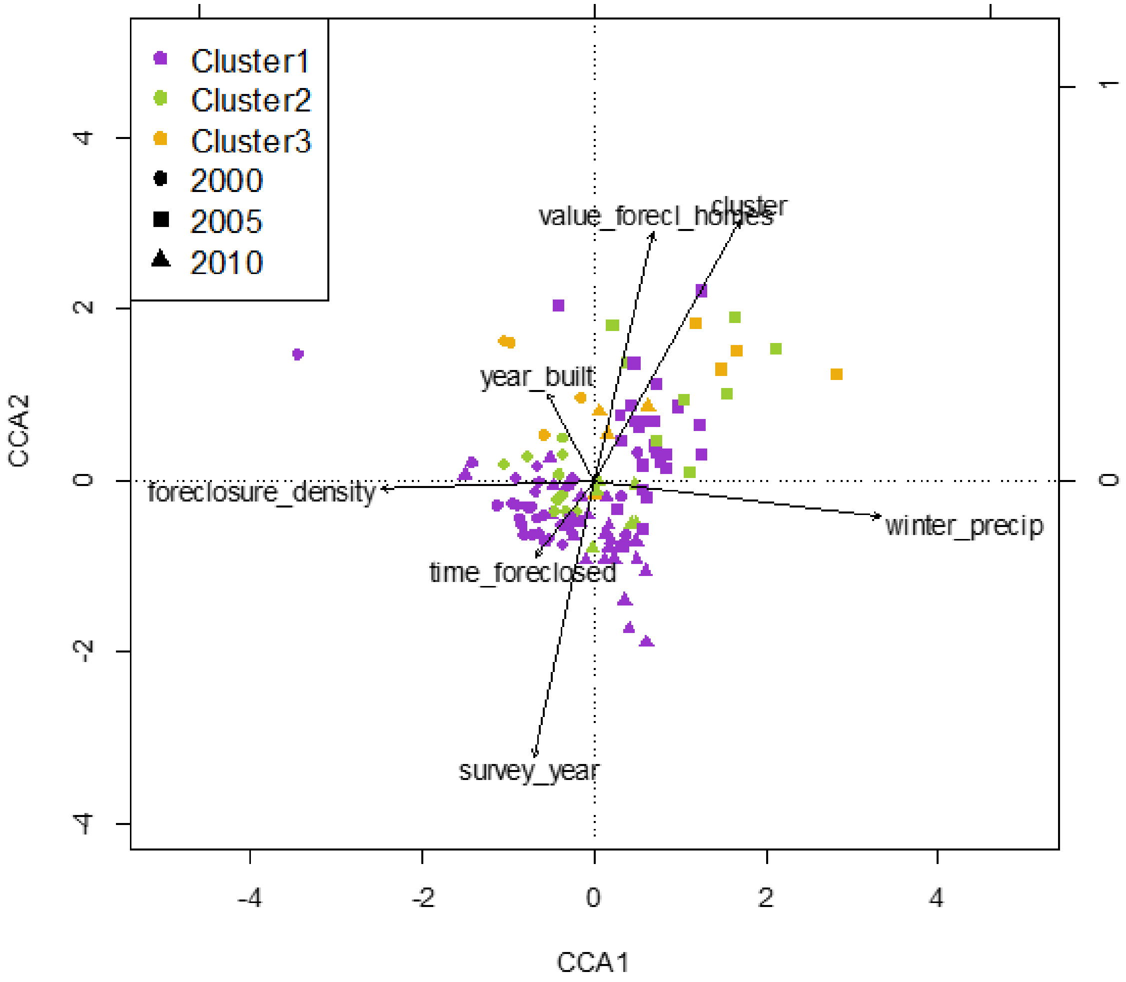

Figure 2.1. Canonical correspondence analysis (CCA) of residential plots as structured by biophysical and socioeconomic disturbance variables. Ordination shows relationship among plots based on similarity in plant species composition. Cluster membership derived from cluster analysis of plots in each year separately are indicated by symbol color and shape.

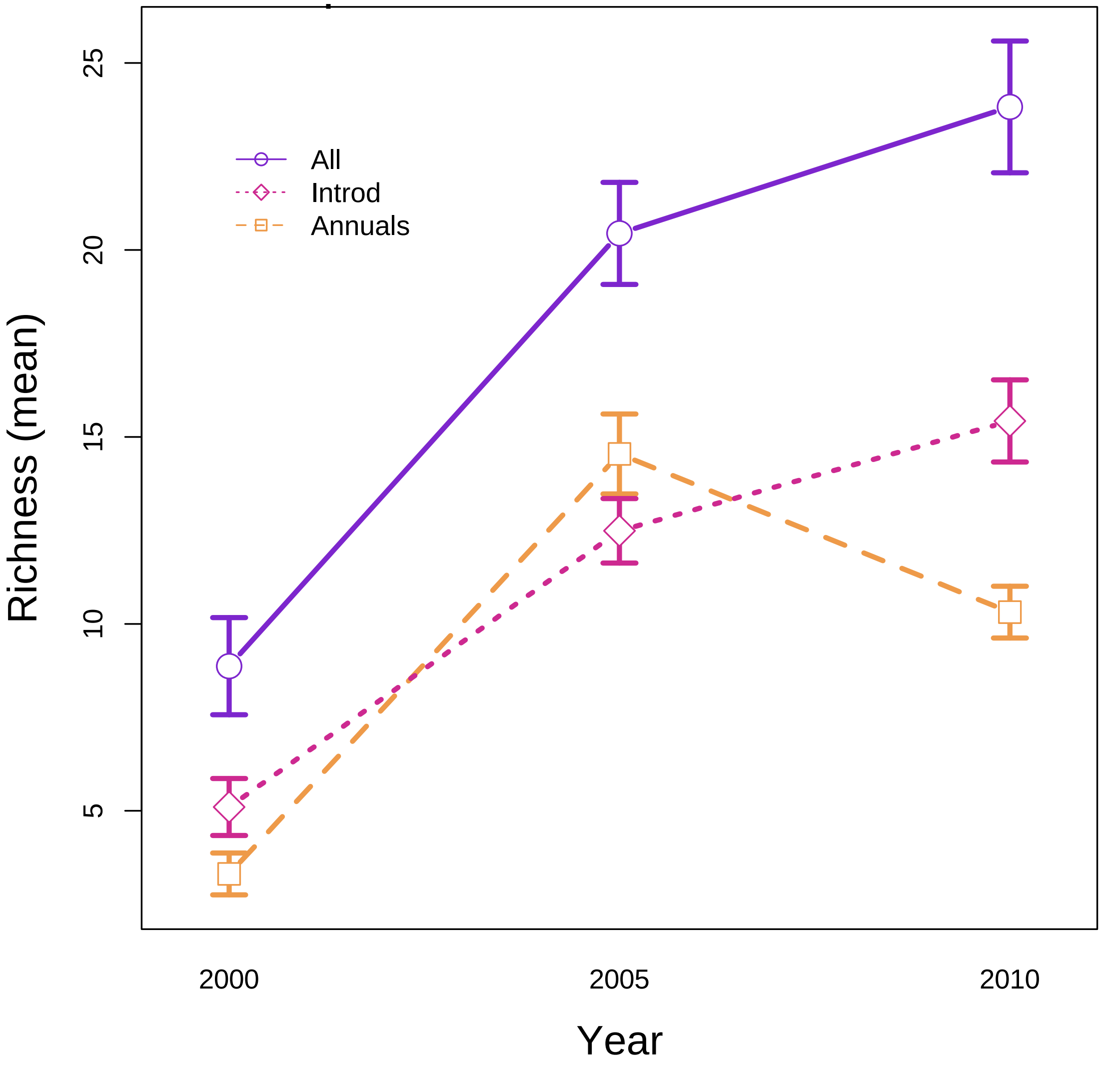

Figure 2.2. Site-level plant species richness of growth forms by provenance and longevity. Points are mean survey site richness values. Values given for pre-recession (2000, 2005) to post-recession (2010). Confidence intervals (95%) shown by error bars.

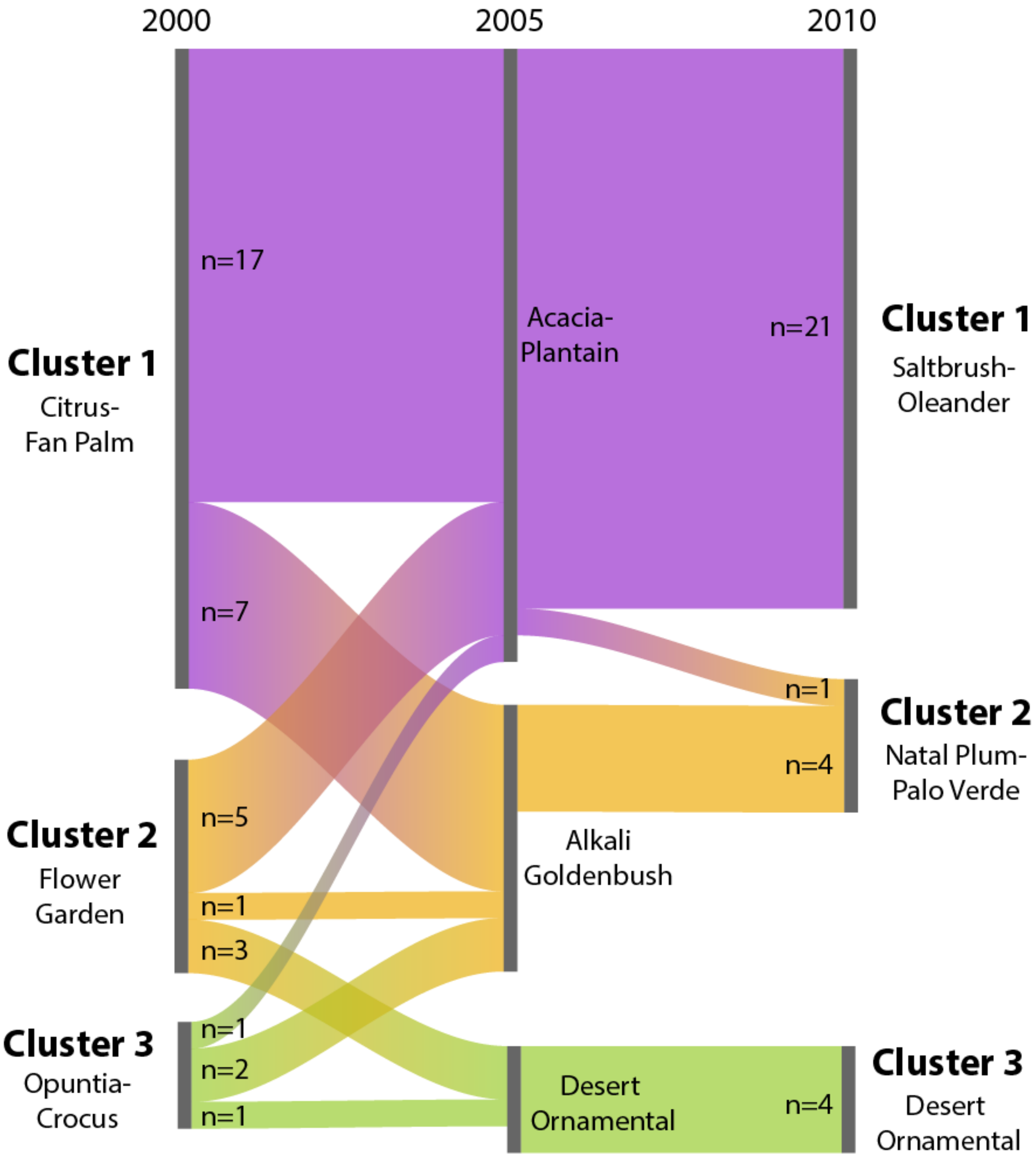

Figure 2.3. Graphical representation of site transitions between differing hierarchical clusters from 2000 through 2010. Grey line width corresponds to the number of plots transitioning among clusters across survey years. Colored boxes denote cluster name and dominant species.

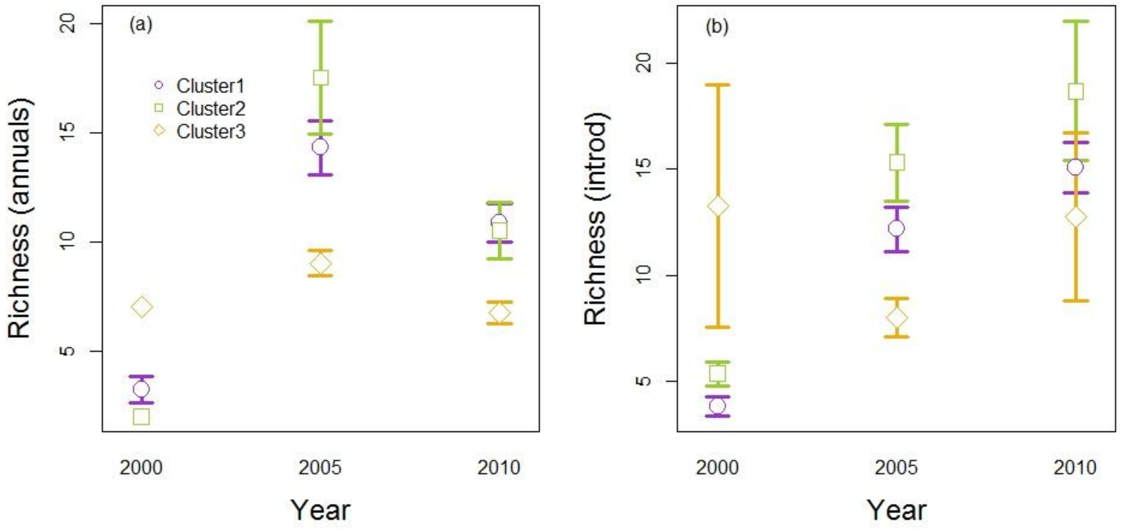

Figure 2.4. Plant species richness of growth forms for hierarchically assigned clusters. Changes in plant species richness shown for (a) introduced species only, and (b) annual plant species only. Values given for pre-recession (2000, 2005) to post-recession (2010). Confidence intervals (95%) shown by error bars.

CHAPTER 3

RESIDENTIAL LANDSCAPING SHIFTS UNDER CLIMATE CHANGE: ARE CULTURAL NORMS ENOUGH TO TRIGGER LARGE-SCALE TRANSITIONS TO LOW WATER USE VEGETATION?

ABSTRACT

Residential areas are the most extensive land cover type in many urban ecosystems, and individual landscaping decisions scale up to affect vegetation cover at the metropolitan level. People make decisions under a complex suite of psychological, social, and economic influences. We developed an agent-based model to examine the relative strength and interaction of these influences. We show how cultural norms lead to large-scale conversion of residential landscaping, and suggest that intervention by local governance may be unnecessary to accelerate residential landscape conversion under the press of climate change.

INTRODUCTION

Residential neighborhoods make up a large proportion of the land-area in many cities – especially in metropolitan areas with sprawling suburban areas and a profusion of single-family homes. For example, in the city of Phoenix, Arizona, 30% of the metropolitan area and 70% of the parcels are zoned as residential (Kane et al. 2014). Residential yards are an important place where individual decisions have a cumulative influence on the larger urban environment. These decisions may seem inconsequential at the household level, but as they assimilate into the urban fabric they can have unintended consequences, like nitrate leaching into groundwater (Milesi et al. 2005), wildlife deaths from rodenticides (Morzillo and Schwartz 2011), or even increased bird diversity (Lerman et al. 2012, Belaire et al. 2014). Landscaping choices affect biodiversity (Bang et al. 2012, La Sorte et al. 2014), water use (Cook et al. 2012), human well-being (Shwartz et al. 2014), and primary productivity (Buyantuyev and Wu 2012). Not unlike urban forests, neighborhood vegetation can be thought of as a common good (Ostrom 1990, Ostrom et al. 1999), that is, shared by and beneficial to the neighborhood and larger regional community.

The spatial heterogeneity of social-ecological patterns and processes complicates our understanding of drivers of vegetation patterns in urban social-ecological systems (SES) (Luck and Wu 2002). Since experimentation is not always an option in SES, simulation modeling is a suitable alternative that allows us to conduct virtual experiments that merge theories from social and ecological sciences (Grimm et al. 2005). Here, we combined economic theory and psychology with climate change adaptation in an agent-based model using a framework of institutional diversity (Ostrom 2005). The institutions framework focuses on how different combinations of rules can generate unexpected outcomes, highlighting the importance of understanding the influence of human behavior.

Climate-induced water shortages have increased in recent years and have recently plagued highly populated areas. The recent California drought, an uncharacteristically hot drought indicative of California's predicted changing climate, highlights the immediacy of the press of climate change, and has reiterated just how vulnerable our global water resources are to climate change and growing populations (Vorosmarty et al. 2000, Diffenbaugh et al. 2015). Under ongoing drought caused by higher than average temperatures and lower than average precipitation, local municipalities are strained to provide water to their residents, and in some cases citations and progressive fines have been implemented to curb wasteful water use and encourage water conserving landscaping choices (Meyer et al. 2014). Although it is difficult to accurately predict extreme drought events or to know whether regions are experiencing a shift in their climate state, there is certainly value in understanding the large-scale effects of human land use decisions in the face of climate-induced biophysical changes.

Cultural norms are rarely considered when it comes to water management decisions (but see Wutich et al. 2012), but in the 1960s Schelling (1969) showed how the simple preference for similarity within a neighborhood could change neighborhood configurations. These cultural norms – informal institutions we define as shared expectations without enforcement mechanisms (Ostrom 1990) – can have powerful effects on people’s decisions that then play out on the larger region. Neighborhood cultural norms exist for yard landscaping. For example, overgrown weeds are considered unsightly and are often thought to decrease property values throughout the neighborhood. In response to an overgrown yard the neighbors may confront the offenders, they may mistreat the offenders and attempt to turn other neighbors against them, or they may escalate the matter and call the local authorities to report the infraction (Nassauer et al. 2009). The authorities could include a management company, homeowners association, or the local municipal government. In any case, these measures may be effective in causing the offending neighbor’s behavior to conform to expectations.

Local governance also has a role to play in people’s yard decisions. In arid and semi-arid cities landscaping conversion programs that offer rebates and incentives for replacing lawns with lower water use landscapes are a tool of governance (Brent et al. In press), and some municipalities educate residents about optimal landscaping for local climate or soils. Within the Phoenix metropolitan areas, some programs offer free desert-adapted trees as incentive to attend the educational programs. But these programs are less rule-oriented and more like nudges in the right direction – providing guidelines that help promote the desired changes in household decisions regarding landscaping and water-use. Rules come into play more when households go against regulations for landscaping decisions or water use in yards. Disincentives such as fines can used by local municipalities or from homeowners associations to discourage certain behaviors.

Many factors influence people’s everyday decisions. In particular, many fields of study are trying to understand how positive psychology and behavioral economics predict consumption behavior. Rational choice theory assumes that individuals act consistently in their choices with regard to preference (Coleman 1990). Since individuals are self-interested, penalties may be required to discourage undesirable behavior and consequently promote a choice less desirable to some individuals (Boyd and Richerson 1992), e.g. water-conserving landscaping practices. The necessity of penalties stands in contrast to the idea that common goods can be managed through informal institutions, that is, cultural norms (Ostrom 1990).

Research questions

By using a complex adaptive systems approach (Holland 1992), we were able to explore the capacity of local institutions (i.e. cultural norms, rules of local governance) to increase water-wise landscaping choices under growing climate stress. Using an agent-based model, we examined how macroscale (regional) patterns emerged from behavioral rules at the microscale (individual agents). We asked: (1) How strong is the role of cultural **norms** in initiating landscaping changes in the face of water stress from climate change? (2) Are formal institutions (i.e. **rules**, penalties) necessary to initiate regional-scale landscaping changes? And, (3) How does **population density** affect the perpetuation of landscaping change throughout a region?

METHODS

This paper explores the problem of how decisions about landscaping yards change vegetation at the regional-level. Interacting household-level components at the local scale lead to emergent phenomena at the macro-level regional scale. We developed an agent-based model (ABM) to explore this question of how decisions about landscaping yards change vegetation at the regional-level because this method allowed us to conduct modeling experiments where we would otherwise be unable to conduct manipulative experiments (Grimm et al. 2005), like in human-dominated systems. ABMs are useful tools for examining emergent phenomena at a macro-level, such as regional vegetation patterns, stemming from the interactions of local-level components, such as household-level decisions about yard landscaping. Pattern-oriented studies have their roots in landscape ecology, where spatial heterogeneity results from interacting ecosystems across multiple scales (Turner 2010). Heterogeneity in initial virtual landscape configuration also contributes to the complexity of the final configuration.

Two contrasting types of residential landscape are found in arid and semi-arid cities throughout the U.S. and elsewhere. Mesic yards consist of irrigated lawns, high water-use plants, and tend not to have gravel or bark mulch cover (Fig. 1); Xeric yards usually consist of drip-irrigated plants, gravel or bark mulch, and do not have lawn cover (Fig. 1) (Martin et al. 2003, Ripplinger et al. In review), but these landscapes can also be mostly bare soil with scant vegetation.

Agent-based model design

A 3-way factorial design was used to implement the ABM experiments (Fig. 2). ABMs featured four key factors. First, in its simplest form, this model examined how neighbors influence individual landscaping decisions. The influence of neighbors is most apparent in front yards, where a household's public countenance is influenced by the cultural norms of the neighborhood. Another way of thinking about cultural norms is as 'keeping up with the Joneses', because people may care deeply about their standard of living in relation to their peers, which in this case results in changing their landscaping. Imitating neighbors is a form of social learning that both signifies and encourages the development of a common culture in the neighborhood (Grove et al. 2014).

Second, water availability influences landscaping decisions either through the cost of water to households, or through availability of precipitation or irrigation water under drought conditions. It is the latter climate influence that we examined here. Third, personal preferences influence choices about landscaping. For example, in the Phoenix metropolitan area it is known that long-time residents are more likely to prefer the lush green lawns and broadleaf trees of traditional mesic landscapes, while more recent settlers to the area are more likely to prefer the xeric landscaping aesthetic that mimics the Sonoran Desert scenery (Larson et al. 2009). Additionally, the importance of neighbor influence should logically vary with population density (Fig. 2), where a lower density population should result in slower perpetuation of landscaping change and higher density result in faster change. And fourth, we examined the effect of formal institutions (i.e. rules) enforced by a penalty from the local municipality.

The purpose of this model is to understand how the behavior of thousands of individual households affects regional-level vegetation dynamics and land cover depending on climate stress, policies of local governance (i.e. municipality rules), and population density. It follows changes in land cover and changes in agent behavior. The main agent in this model is a household that makes decisions about land cover conversions on a monthly basis. A household is the smallest unit in this ABM. The key properties associated with each household-agent in the model are: (1) Preference of agent for neighbors with similar landscaping. Neighbor preference changes to match agent's current type of landscaping. (2) Agent's ability to resist changing to a non-preferred type of landscaping, as determined by an 'energy' level to resist climate pressure to change landscaping. Initial energy value represents an agent's ability to resist up to 12 months of drought stress. In the scenarios involving a penalty for remaining in water-intensive mesic landscaping, agents also have (3) a tolerance threshold representing the agent's ability to remain in mesic landscaping. The penalty tolerance value represents an agent's ability to pay fines. Though the assigned penalty tolerance value was selected through trial and error, it can be thought of as relating to a household-agent's wealth. Each simulation timestep is equivalent to 1 month. Starting values for agent properties (2) and (3) are given in Table 1.

In the ABM, municipalities use penalties to enforce water-conserving landscaping practices. The model includes two phases. (1) In the initialization phase the model structure is created, including the establishment of household-agents and the modeled urban grid in which they interact. Half the household-agents are assigned mesic yards and the other half xeric yards. Individual household-agents are randomly assigned to a location in the square grid representing the urban region, using random initial conditions as a basis for household assignments. Each agent is allotted a randomized amount of starting energy to resist climate pressure to change to a xeric landscape under water stress (Table 1). (2) In the simulation phase several processes are repeated in each simulation period. First, the household-agent looks at its 8 adjacent neighbors and decides whether it is satisfied with its landscaping type (mesic or xeric), then changes its landscaping if there is sufficient pressure from neighbors to be similar to them. Then, if the amount of pressure due to climate stress from drought is higher than the agent's energy remaining to resist change, the agent changes its landscaping mode to xeric (Table 1). Each agent's energy to resist is reduced by climate pressure with every timestep of pressure above their resistance energy level. Such that as climate pressure accumulates for each individual agent with a mesic yard, their resistance energy decreases with each timestep. Once the agent has no more resistance energy, they change their landscaping to a xeric yard. Simulations were run using five different levels of climate pressure. Climate pressure levels were designed to relate to the hypothetical drought stress agents experience as climate change worsens – 'Very Low' through 'Very High' climate stress, where 'Very Low' optimistically represents current climate stress and 'Very High' represents a worst-case climate scenario (Fig. 2).

In the simulations that involve a penalty for remaining in a water-intensive mesic landscape, in addition to the above simulation procedure, a mesic household-agent receives a penalty that accumulates additively with each timestep. Once the cumulative penalty reaches the individual agent's threshold, the agent is unable to afford more fines and subsequently changes to a xeric landscape (Table 1). Finally, at the end of each simulation timestep the household-agents update their landscaping mode, their energy levels to resist, their satisfaction with adjacent landscapes, and their cumulative penalties. Each simulation was run for up to 60 timesteps (roughly equivalent to 5 years), for varying levels of population density. Modeling was implemented using the NetLogo software interface version 5.2.0. NetLogo interface and example are shown in Fig. 3.

We designed a model that allowed us to conceptually investigate future climate change scenarios where water provisioning is reduced. There were two contrasting scenarios, one where cultural norms and climate pressure induced changes in the region, and the other where a penalty or fine was imposed by local governance to promote landscaping conversions across the region in addition to the pressure of norms and climate change. The assumption motivating the second scenario was that a penalty was necessary to accelerate the rate of change towards a region dominated by xeric landscaping, but it has been shown that common pool resources – like urban vegetation – can be governed by people who use the landscapes rather than needing government intervention (Ostrom 1990, Ostrom et al. 1999). We explored these contrasting hypotheses by comparing the outcomes of the two scenarios (Fig. 2).

RESULTS

With our agent-based model, we explored the first scenario – cultural norms and climate pressure, without a penalty enforcing xeric landscaping. Cultural norms under climate pressure had a strong and rapid effect on all but the lowest climate pressure scenario, and the rate of the response increased with the strength of the climate pressure

(Fig. 4a). These findings lend support to ideas of common-pool resource governance (Ostrom 1990, Ostrom et al. 1999), giving the impression that large-scale residential landscaping can be shifted towards a very high proportion of water-conserving xeric landscapes by cultural norms alone under climate pressure.

Next, we explored the scenario that included the addition of a penalty from local governance to promote conversions to xeric landscaping. We found that the addition of a penalty for having a water-hungry, mesic landscape had a negligible effect on the regional percentage of xeric landscapes in the short-term, but in the long-term the highest penalty only marginally augmented the percentage of xeric landscapes (Fig. 4b). So contrary to the starting assumption that a penalty would accelerate the rate of change in landscaping regionally, instead we found that cultural norms under climate change pressure induced a more rapid response than even the highest penalty.

We explored the effect of varying population density on the outcomes of these two scenarios. We found that the magnitude of a shift to xeric landscaping region-wide is dependent on population density (Fig. 5). In the simpler scenario, the lowest population density resulted in the smallest percentage of xeric landscapes, and the highest population density resulted in the highest percentage of xeric landscapes across the region (Fig. 6a). In the scenario that includes the landscaping penalty, mid and high population densities resulted in xeric percentages lower than the lowest population density, followed by a spike to in the highest population density scenario (Fig. 6b). It is plausible that at the lowest population density, the shift to xeric landscaping was perpetuated through climate and penalty effects (which are not density dependent). In the mid and high density scenarios agents may have been more reluctant to change because the effects of cultural norms were density dependent and neighbors factored in more highly, which resulted in the agents struggling to decide whether or not to change landscaping despite climate and penalty effects.

DISCUSSION

By applying our knowledge of the effects of institutions (i.e. rules, norms) on this common good, we were able to assess potential effects of individual decisions on regional land cover. Neighborhood vegetation is known to have positive effects on biodiversity, carbon sequestration, water quality maintenance, and other urban ecosystem services important to human well-being (Peterson et al. 2010, Eigenbrod et al. 2011). According to Ostrom's (1990) ideas of common-pool resource management, shared resources like neighborhood vegetation can be successfully and freely governed by people who use them rather than needing government intervention. Our first scenario supports this idea by showing that rapid response to climate pressure emerged without the intervention of penalties included in the second scenario. Notably, this occurred in modelled cities with Very High density housing (100% of the patches are filled by residential housing), where we expected higher density to correlate to faster response to neighborhood norms.

The highest population density resulted in the fastest response to climate pressure on vegetation in the first scenario (no penalties). This may give us insights into how to achieve regional goals for residential vegetation by handling higher population urban areas differently than lower density rural areas. If the mechanisms driving regional landscaping shifts in densely populated areas include cultural norms, penalties like fines for mesic landscapes may not be necessary. In lower density areas, on the other hand, penalties may be highly effective.

Local governance may use penalties to enforce landscaping practices, acting as an architect of ‘beneficent’ choices by creating an environment that encourages particular outcomes. However, it can be difficult for non-profit utility companies to implement penalties like charging fines for infractions, because fines result in surplus revenues. Another option may be to use social comparisons to encourage certain behaviors – e.g. “Your neighbor received a $2000 incentive for changing their lawn to a water-wise landscape”. The use of social comparisons has been successful in changing water consumption (Ferraro and Price 2013), which is a way of using social norms to shift individual behavior toward conservation behaviors without the complication of implementing a fine. Another way local governance can steer behavior is by considering how the choice is framed for new residents. What people decide they want is often affected by the way the choice is framed for them (Levin et al. 1998). If new residents are automatically enrolled in a landscape conversion plan when setting up their utilities, they would ‘opt in’ to the water-use mandates of the municipality. It simplifies their decision, and because they’ve chosen it based on the way it was framed (e.g. information provided when buying/renting, attractive xeric landscapes), they are more likely to be satisfied with it.

As human influence increases on ecosystems across the globe (Foley et al. 2005), humankind is confronted with the need to advance and implement understanding of urban governance. Improved urban governance can mitigate some of the local decreases in ecosystem services. If a municipality’s goal is to promote the conversion or adoption of water-conserving landscaping, local cultural norms may be more effective than formal institutional rules of local governance, particularly in high population density areas and under more extreme water stress. If the goal is to work within existing local governance, fines can be an effective way of prompting landscape conversion, though it may remain more effective to allow collective action to take the reins.

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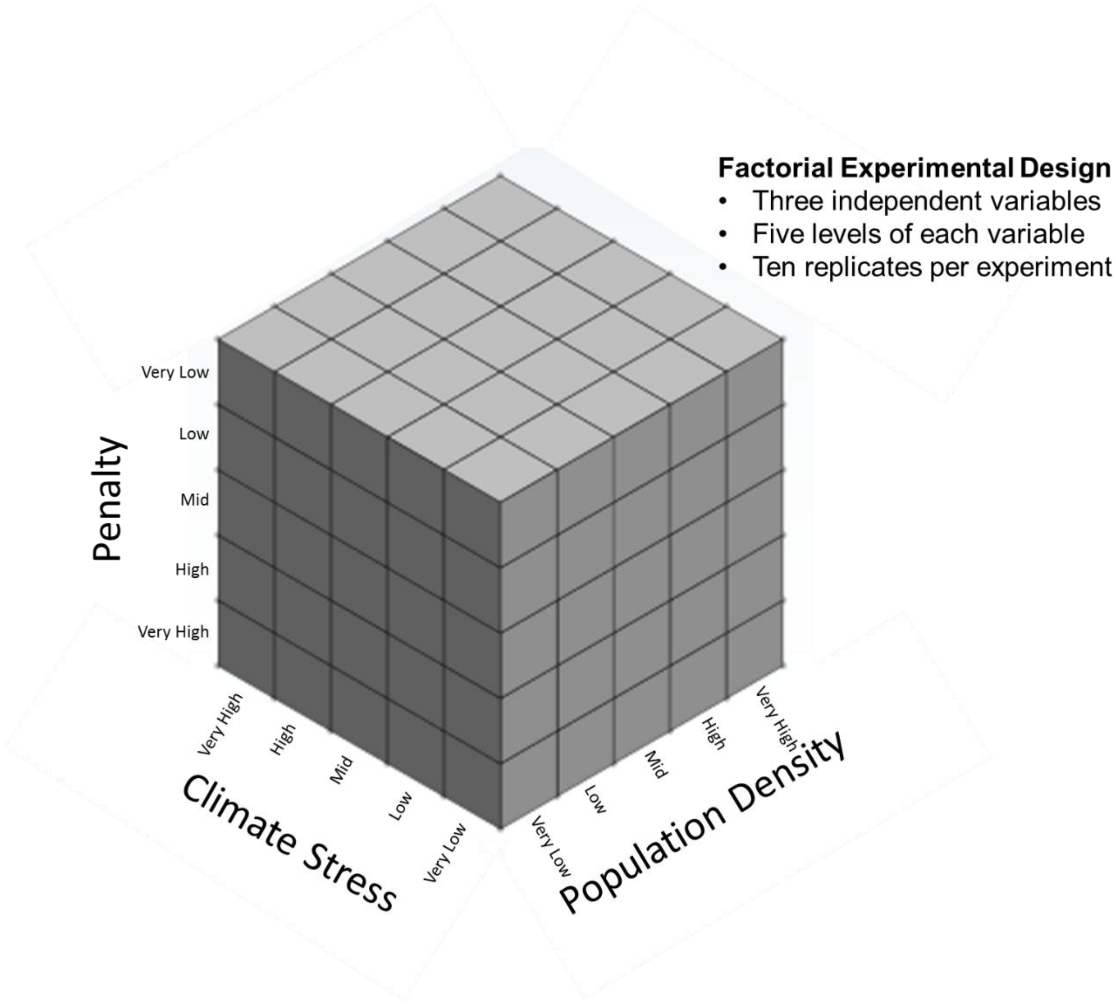
Table 3.1. ABM rules and variables.

Properties	Description	Initial Value	Rule	Action
Landscaping preference	Agent preference for neighbors to be similar to self	62.5%	If the number of similar neighbors you have is **greater** than or equal to what you wanted (5 of 8 or more)…	…No change to landscaping.
			If the number of similar neighbors you have is **less** than what you wanted (less than 5 of 8 similar)…	…Change to other type of landscaping.
Energy to resist landscaping change	Ability to resist climate pressure (aka landscaping preference strength)	random number between 0 and 12	Every timestep, subtract the value of "climate pressure" (0-10)	Once resistance energy level reaches 0, change to xeric landscape.
Penalty tolerance	Amount of penalty accumulated for having mesic landscaping	0	Every timestep landscape is not xeric, add 0.1	Once fine reaches individual-agent's threshold of 36 ± 12, change to xeric landscape.

Figure 3.1. Example of a mesic or high water-use landscape (left) and a xeric or water-conserving landscape (right) in Phoenix, Arizona.



Factorial Experimental Design

- Three independent variables
- Five levels of each variable
- Ten replicates per experiment

Figure. 3.2. Experimental design. A 3-way factorial experiment, with five factor levels each. Ten replicates were run for each experimental combination.

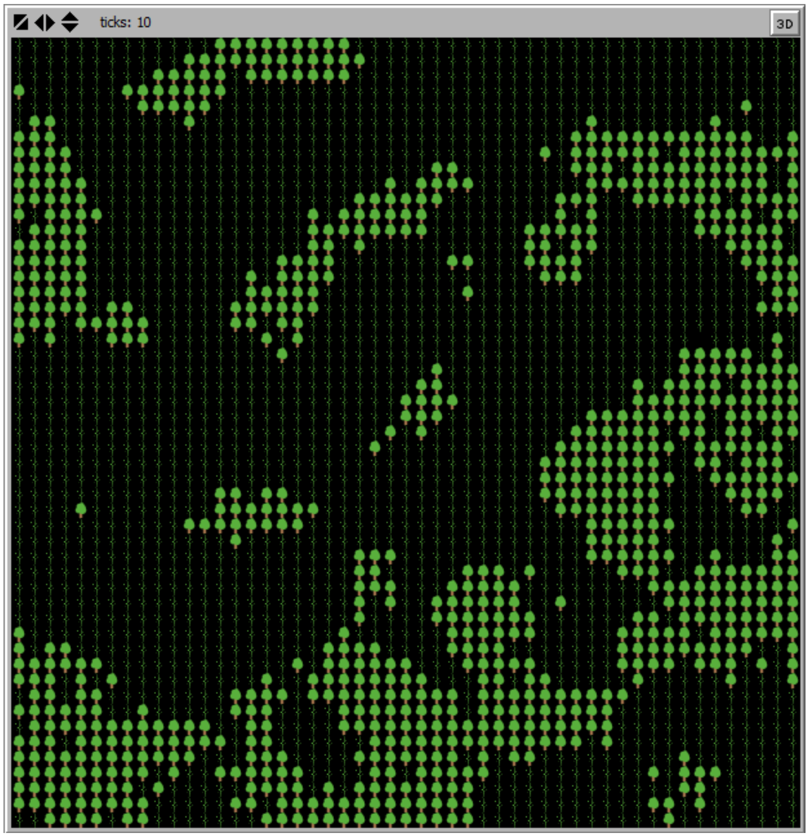


Figure 3.3. Example of the NetLogo modeling interface. This image was captured mid-simulation for a Very High population density, Low climate pressure, High mesic landscape penalty scenario. Green tree icons represent agents with mesic landscapes, and thinner cactus icons represent agents with xeric landscapes.

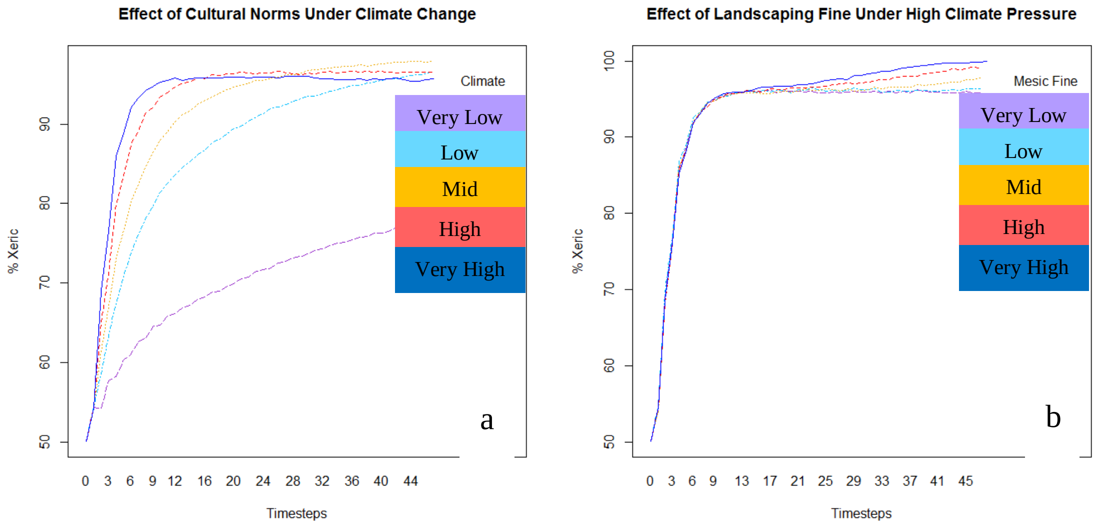


Figure 3.4. Change in the percentage of household agents with xeric landscapes, over time, for simulations that varied by the amount of pressure from climate warming (aka decreased water provisioning). Panel (a) shows model results for climate pressure ranging from Very Low to Very High, under only the pressure of cultural norms and agent landscaping preference. Panel (b) shows results for the same model but with the addition of water-conserving landscaping enforced via fines from local governance. Panel (b) results are for scenarios of climate pressure fixed at Very High levels, under the additional pressure of penalty for mesic landscaping, ranging from Very Low to Very High. Results shown are the mean of 10 simulations at each level, and are for a population density of 100% (when all of the patches are residential housing).

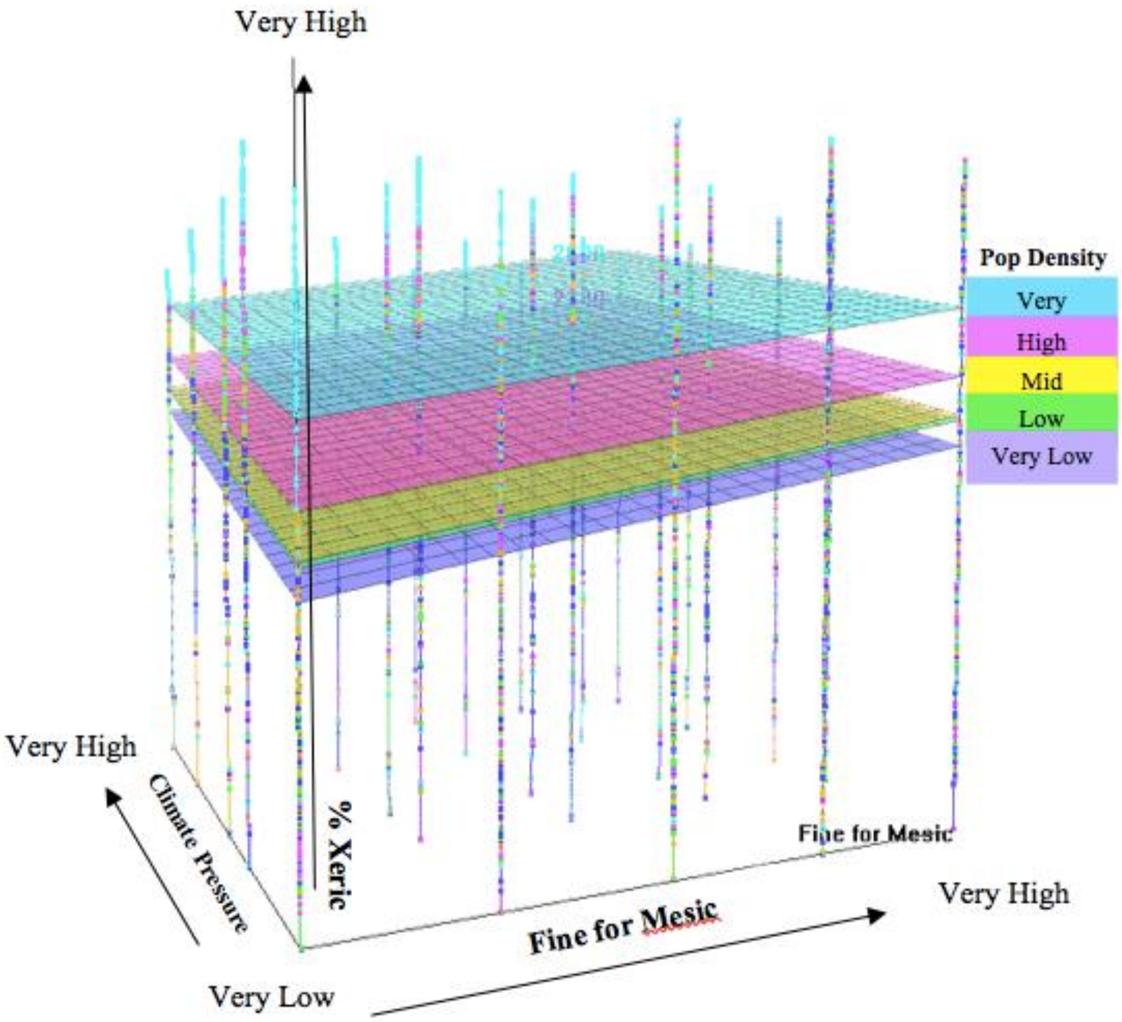


Figure 3.5. Xeric outcomes (z-axis) for model runs examining a range of increasing climate pressure (x-axis), a range of increasing fines for mesic landscapes (y-axis), and a range of population densities (point color and regression planes). Linear regression surface planes fitted to model outcomes.

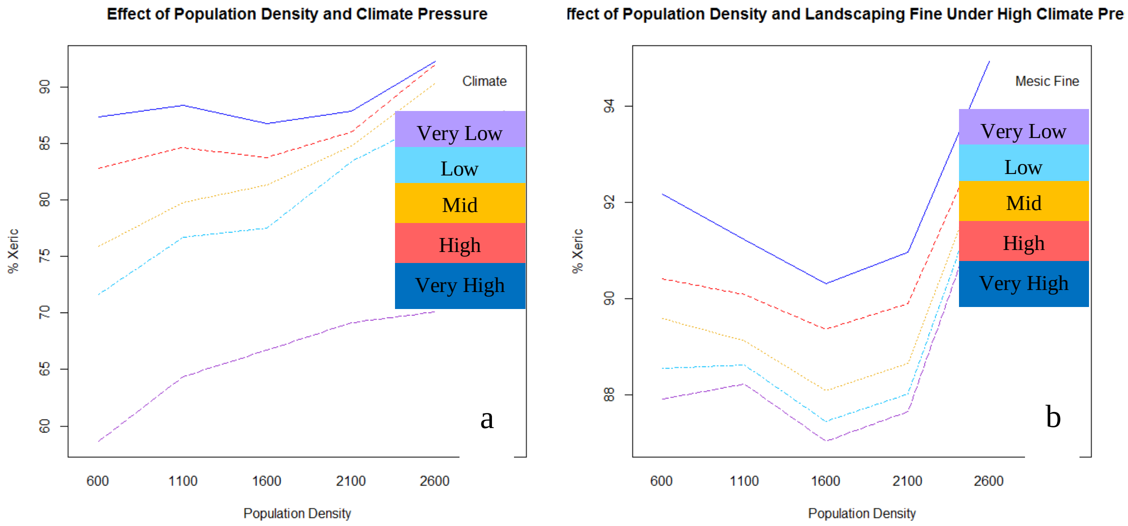


Figure 3.6. Change in the percentage of household agents with xeric landscapes, by population density and the amount of pressure from (a) climate warming and from (b) climate warming plus governance. Panel (a) shows population density results for the pressure of cultural norms and increasing pressure from climate warming. Panel (b) shows population density results over increasing landscaping penalties for Very High climate pressure.

Chapter 4

CONCLUDING REMARKS

The earth's vegetation has been radically altered from local to global scales, with urbanization accelerating land cover change and threatening the sustainability of the ecosystems within which most of humanity resides (Vitousek et al. 1997, Foley et al. 2005). Scholars have made progress towards understanding spatial vegetation patterns of cities and in identifying relevant drivers of those patterns – drivers that include socioeconomic status and land use legacies (Grove et al. 2006, Hope et al. 2006, Walker et al. 2009, Boone et al. 2010, Buyantuyev and Wu 2012, Johnson et al. 2015). However, research has not yet addressed the dynamic context that sets the stage for interacting effects of socioeconomic fluctuations, global change, and human decisions on urban vegetation. With a better understanding of urban vegetation dynamics and drivers, policymakers can consider vegetation response when addressing issues of ecosystem function, environmental justice, and urban heat island effects.

Here, I addressed this gap by considering the temporal dynamics of urban vegetation using existing long-term data and also simulation modeling. With these tools, I evaluated dynamics of vegetation community and cover, and examined the relative importance of biophysical, institutional, and socioeconomic drivers.

In this concluding section, I highlight the main contributions from each chapter and synthesize the major findings of my doctoral research.

Chapter 1: By examining a multiscale time series of plant communities in the Phoenix metropolitan area, I found that plant species diversity increased over time from 2000 to 2010 at both residential and regional scales. Composition of the species associated with agricultural and urban plots – both highly managed types of land cover – became more similar with time. This compositional shift, referred to as ecological or biotic homogenization, is suggestive of the prominent role of human preferences in shaping designed ecosystems. Species composition on residential plots, however, became more heterogeneous among three prevalent types of residential landscaping with time, possibly as a result of unequal effects of the Great Recession on homeowners with preferences for varied types of landscaping.

Chapter 2: The recent housing boom and subsequent Great Recession (2007-2010) provided a natural experiment to determine the effects of socioeconomic fluctuations on residential landscapes. I identified socioeconomic and biophysical variables controlling the direction of changing residential vegetation. Increases in overall species richness were controlled by the development associated with the housing boom, as were changes in the composition of species found on residential plots. Results supported my expectation that the number and kinds of landscaping plants used in a booming economy would be different than during a stable or failing economy. Annual plant species were linked to winter precipitation, demonstrating similar behavior to native desert plant communities. Between 2005 and 2010 species composition changed on residential plots, but sites with similar composition changed in the same direction, further demonstrating the indirect but strong effects of the socioeconomic disturbance to people’s yards.

Chapter 3: Examining the long-term effects of different landscaping management scenarios under climate change, I demonstrated that under the right circumstances cultural norms can lead to large-scale regional conversions of residential landscaping to drought tolerant vegetation. Local governing bodies may consider the policy option of penalizing offending households for high water-use landscaping, but this type of intervention may be unnecessary to hasten residential landscaping change. Penalties accelerated the conversion process, but were most effective in lower density regions where the effectiveness of cultural norms is reduced by diminished contact with neighbors.

This dissertation used a flexible conceptual framework to integrate and examine the effects of biophysical processes, anthropogenic activities, disturbance, and socio-political context on urban vegetation dynamics (Fig. 0.1). Overall, my findings highlight the importance of understanding people's responses to the economy, boom-bust cycles in the housing market, and cultural pressures on how urban landscapes are managed and designed. This complex suite of factors influencing urban vegetation dynamics presents urban planners and policy makers with many points of entry into managing urban vegetation and its associated ecosystem services. My research also highlights challenges to understanding the factors influencing urban ecosystem functioning. These findings and the conceptual framework enhance our understanding of complex elements acting on urban vegetation and contribute to a socioecological approach to managing urban landscapes for the many facets of biodiversity and ecosystem functioning.

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APPENDIX A

TABLE S1.1. ACCOMPANIES CHAPTER 1. COMPLETE LIST OF SPECIES.

Table S1.1. Complete species list with species Importance Values (IVs) for each land-use land cover (LULC) type surveyed in 2010.

	Species Importance Values (IVs)					
	Agri	**Desert**	**Urban**	**Mesic**	**Oasis**	**Xeric**
Acacia	**0**	**2**	**1.62**	**0**	**1**	**2.67**
Acacia constricta	**0**	**2.14**	**1.5**	**0**	**0**	**2**
Acacia farnesiana	**0**	**0**	**2.25**	**6**	**0**	**1**
Acacia salicina	**0**	**0**	**1**	**1**	**1**	**1**
Acacia saligna	**0**	**0**	**2**	**0**	**0**	**2**
Acacia stenophylla	**0**	**0**	**2.5**	**0**	**3**	**0**
Agave	**0**	**0**	**2.77**	**6**	**2.5**	**2.33**
Agave americana	**0**	**0**	**1.88**	**1**	**2**	**3**
Agave desmetiana	**0**	**0**	**1**	**1**	**0**	**0**
Agave ferdinandi regis	**0**	**0**	**1**	**1**	**0**	**0**
Agave geminiflora	**0**	**0**	**2**	**0**	**2**	**2**
Agave macroacantha	**0**	**0**	**2**	**0**	**2**	**0**
Agave schidigera	**0**	**0**	**1**	**1**	**0**	**0**
Agave vilmoriniana	**0**	**0**	**1.5**	**1**	**2**	**0**
Allium sativum	**0**	**0**	**2**	**0**	**2**	**0**
Aloe	**0**	**0**	**1.5**	**3**	**1**	**1.25**
Aloe variegata	**0**	**0**	**1**	**0**	**0**	**1**
Aloe vera	**0**	**0**	**2.75**	**4**	**3**	**3**
Aloysia wrightii	**0**	**0**	**1**	**0**	**1**	**0**
Amaranthus albus	**1**	**0**	**1**	**0**	**1**	**0**
Amaranthus blitoides	**1**	**0**	**1**	**0**	**0**	**1**
Ambrosia deltoidea	**0**	**4.51**	**3.5**	**0**	**0**	**1**
Amsinckia	**0**	**1**	**1**	**0**	**1**	**0**
Amsinckia menziesii	**1.5**	**1.19**	**1.14**	**0**	**0**	**1**
Ananas comosus	**0**	**0**	**1**	**0**	**1**	**0**
Antigonon leptopus	**0**	**0**	**4**	**0**	**4**	**0**
Aptenia cordifolia	**0**	**0**	**1**	**0**	**0**	**1**
Arecaceae	**0**	**0**	**1.5**	**0**	**0**	**2**
Aristida purpurea	**0**	**2.67**	**1**	**0**	**1**	**1**
Artemisia filifolia	**0**	**0**	**1**	**0**	**1**	**0**
Asparagus densiflorus	**0**	**0**	**2.57**	**2**	**4**	**0**
Asparagus setaceus	**0**	**0**	**2**	**2**	**0**	**0**
Asteraceae	**0**	**1.2**	**1**	**1**	**1**	**1**
Astragalus nuttallianus	**0**	**1**	**1.25**	**1**	**0**	**0**
Atriplex canescens	**0**	**1**	**3**	**0**	**0**	**5**

Atriplex elegans	1.33	1	1.62	2.67	1	1.33
Atriplex wrightii	0	0	1	0	1	0
Avena fatua	1	1	1	0	1	0
Baccharis	0	3	2	0	2	0
Baccharis sarothroides	4	5	2.12	2.25	1.4	1.71
Baileya multiradiata	0	1	1.67	1	0	1
Bauhinia variegata	0	0	2	2	0	0
Beaucarnea recurvata	0	0	6	6	0	0
Boerhavia coccinea	0	0	1	1	1	1
Boerhavia intermedia	0	0	1	0	1	0
Boraginaceae	0	1	1	1	1	0
Bougainvillea	0	0	2.38	2.75	2.5	2
Bougainvillea spectabilis	0	0	2.5	0	3	1
Brachychiton	0	0	1	0	1	0
Brachychiton populneus	0	0	2.75	4	3.5	0
Brahea edulis	0	0	2	2	0	0
Brassica tournefortii	0	1.08	1.17	0	0	1
Bromus arizonicus	0	1	1.5	1	0	0
Bromus catharticus	1	0	1	1	1	1
Bromus rubens	0	1.11	1.18	1	0	1
Buxus microphylla	0	0	1	1	0	0
Cactaceae	0	1	2.29	1	2.5	4
Caesalpinia mexicana	0	0	1	0	1	0
Caesalpinia pulcherrima	0	0	2.25	1	1	1.5
Calliandra	0	0	1	0	1	0
Calliandra californica	0	0	1.67	1	1	1
Callistemon viminalis	0	0	5	2	6.25	0
Camellia	0	0	2	0	2	0
Canna	0	0	5	5	0	0
Capsella bursa pastoris	1	0	1.29	1.67	1	1
Carex subgen Vignea	0	0	1	1	1	0
Carissa macrocarpa	0	0	5	5.33	5.5	5
Carnegiea gigantea	0	2.44	1.83	0	1.33	2.5
Carpobrotus edulis	0	0	1	0	0	1
Carya illinoinensis	0	0	2	1	3	0
Catharanthus roseus	0	0	2.43	4	3	1
Caulanthus lasiophyllus	0	1.08	1	1	1	0
Celtis	0	0	3	0	0	3
Cenchrus ciliaris	0	2.5	1	0	0	1
Cenchrus setaceus	0	5	2.67	2	1	4
Ceratonia siliqua	0	0	1.5	0	1	0

Cereus	**0**	**0**	**2.27**	**1**	**3.5**	**3**
Chamaerops humilis	**0**	**0**	**1.4**	**0**	**0**	**2**
Chamaesyce maculata	**0**	**0**	**1.15**	**1**	**1.33**	**1**
Chenopodiaceae	**1**	**0**	**1**	**1**	**1**	**0**
Chenopodium	**1.5**	**1.33**	**1.21**	**1.67**	**1**	**1**
Chenopodium berlandieri	**2**	**0**	**1**	**1**	**1**	**1**
Chenopodium murale	**0**	**1**	**1**	**0**	**0**	**1**
Citrus	**1**	**0**	**3.05**	**3.75**	**4.14**	**2.67**
Citrus limon	**0**	**0**	**1.33**	**1.5**	**1**	**1**
Citrus sinensis	**0**	**0**	**2.44**	**2**	**3.67**	**1**
Citrus x paradisi	**0**	**0**	**1.7**	**2**	**1.5**	**2**
Convolvulus	**0**	**0**	**1**	**0**	**1**	**0**
Convolvulus arvensis	**0**	**0**	**1**	**1**	**1**	**0**
Convolvulus cneorum	**0**	**0**	**2**	**0**	**4**	**1**
Conyza	**1**	**0**	**1.27**	**1.5**	**1.17**	**1**
Conyza bonariensis	**0**	**0**	**1**	**0**	**1**	**1**
Conyza canadensis	**0**	**0**	**1.33**	**1.5**	**0**	**0**
Copiapoa cinerea	**0**	**0**	**1**	**0**	**1**	**0**
Coriandrum sativum	**0**	**0**	**1**	**1**	**0**	**0**
Cotula australis	**0**	**1**	**1**	**1**	**1**	**0**
Crassulaceae	**0**	**0**	**3**	**0**	**3**	**0**
Cryptantha	**0**	**1.17**	**1**	**1**	**0**	**1**
Cryptantha angustifolia	**1**	**1**	**1**	**0**	**1**	**1**
Cryptantha barbigera	**0**	**1**	**1**	**1**	**0**	**0**
Cryptantha maritima	**0**	**1.11**	**1**	**0**	**1**	**0**
Cryptantha micrantha	**0**	**1**	**1**	**0**	**1**	**0**
Cryptantha muricata	**0**	**1**	**1**	**1**	**0**	**1**
Cuphea	**0**	**0**	**5**	**0**	**5**	**0**
Cupressaceae	**0**	**0**	**1.67**	**0**	**3**	**1**
Cupressus sempervirens	**7**	**0**	**2.5**	**2.5**	**0**	**0**
Cycas revoluta	**0**	**0**	**1**	**1**	**1**	**1**
Cyclospermum leptophyllum	**0**	**0**	**1**	**1**	**1**	**1**
Cylindropuntia	**0**	**3.67**	**3**	**0**	**0**	**3.5**
Cylindropuntia acanthocarpa	**0**	**3.41**	**3**	**0**	**0**	**1**
Cylindropuntia bigelovii	**0**	**4**	**4**	**0**	**4**	**0**
Cylindropuntia ramosissima	**0**	**0**	**2**	**0**	**2**	**0**
Cynodon dactylon	**0**	**0**	**1.06**	**1**	**1**	**1.17**
Dalbergia sissoo	**0**	**0**	**2.25**	**5**	**2.67**	**1.5**
Dalea greggii	**0**	**0**	**3.5**	**0**	**0**	**2**
Dasylirion wheeleri	**0**	**0**	**2.58**	**0**	**2.75**	**3**
Datura	**0**	**0**	**1**	**0**	**1**	**0**

Dichondra micrantha	0	0	1	1	1	0
Dimorphotheca sinuata	0	0	1.5	0	1	1
Dolichandra unguis-cati	0	0	1.17	2	1	0
Duranta erecta	0	0	1	1	1	0
Ebenopsis ebano	0	0	2.2	6	1.5	0
Echinocactus	0	0	2	0	2	0
Echinocactus grusonii	0	0	3.62	0	8	4
Echinocereus	0	3.2	2	0	2	0
Echinocereus pectinatus	0	0	1	0	1	0
Echinochloa colona	0	0	1	1	0	0
Echinopsis	0	0	1.5	0	1.5	0
Echinopsis chamaecereus	0	0	1	0	1	0
Encelia farinosa	1	4.29	3.1	0	0	1.5
Eremophila	0	0	1	0	1	0
Eremophila maculata	0	0	2.25	2.33	0	0
Eriogonum	1	1.5	1	0	1	0
Eriogonum deflexum	1	2	1	0	1	1
Erodium cicutarium	1	1.04	1.09	1	1.22	1
Erodium texanum	0	1.08	1	1	0	1
Erythrostemon gilliesii	0	0	1.5	0	1	2
Eucalyptus	0	0	2.25	1	1	6
Eucalyptus microtheca	0	0	4	0	0	4
Euonymus japonicus	0	0	1	0	0	1
Euphorbia	1	1	1.38	1.33	1.8	1
Euphorbia albomarginata	0	1	1.08	1	1.2	1
Euphorbia capitellata	0	1	2	0	2	0
Euphorbia hyssopifolia	0	0	1	0	1	0
Euphorbia micromera	0	1	1	0	0	1
Euphorbia polycarpa var *polycarpa*	0	1	1	0	0	1
Euphorbia prostrata	1	0	1	0	0	1
Euphorbia tirucalli	0	0	1.5	0	2	1
Fabaceae	0	0	1.33	0	1.5	0
Ferocactus	0	2.85	3.29	0	1	0
Ferocactus cylindraceus	0	2.88	1.33	0	1.5	1
Ferocactus wislizeni	0	4	1.71	0	1.67	1.5
Ficus	0	0	2.2	0	1	4
Ficus carica	0	0	4	0	4	0
Ficus microcarpa	0	0	1	0	1	0
Ficus microcarpa var *nitida*	0	0	3.8	2.67	5.5	0
Fouquieria columnaris	0	0	1	0	1	0

Fouquieria splendens	**0**	**3.45**	**1.67**	**0**	**1.5**	**1.75**
Fraxinus	**0**	**0**	**1.8**	**2**	**0**	**0**
Fraxinus pennsylvanica subsp *velutina*	**2**	**0**	**1.73**	**1.75**	**4**	**1.75**
Fraxinus uhdei	**0**	**0**	**2**	**0**	**2**	**0**
Gardenia	**0**	**0**	**1**	**1**	**0**	**0**
Gardenia jasminoides	**0**	**0**	**5**	**9**	**0**	**1**
Glandularia pulchella	**0**	**0**	**1**	**0**	**1**	**0**
Grevillea robusta	**0**	**0**	**1**	**0**	**0**	**1**
Hedypnois cretica	**0**	**0**	**1**	**1**	**1**	**0**
Hemerocallis	**0**	**0**	**5**	**5**	**0**	**0**
Herniaria hirsuta	**0**	**1.08**	**1.12**	**1**	**2**	**1**
Hesperaloe parviflora	**0**	**0**	**2.53**	**5**	**4**	**1.75**
Hibiscus	**0**	**0**	**1**	**1**	**0**	**0**
Hibiscus rosa-sinensis	**0**	**0**	**3.57**	**3.33**	**4**	**0**
Hoffmannseggia glauca	**0**	**0**	**1**	**0**	**0**	**1**
Hordeum murinum	**1.5**	**1**	**1**	**1**	**1**	**1**
Hordeum vulgare	**0**	**0**	**1**	**0**	**1**	**0**
Hyacinthus	**0**	**0**	**1**	**0**	**1**	**0**
Jacaranda mimosifolia	**0**	**0**	**1.67**	**2**	**0**	**0**
Jasminum	**0**	**0**	**1**	**0**	**1**	**0**
Juniperus	**0**	**0**	**1.75**	**2**	**1.5**	**0**
Justicia californica	**0**	**3.25**	**2**	**0**	**2**	**0**
Justicia spicigera	**0**	**0**	**2**	**0**	**2**	**0**
Lactuca serriola	**1.33**	**1**	**1.33**	**1.64**	**1.46**	**1.25**
Laennecia coulteri	**0**	**1**	**1**	**0**	**0**	**1**
Lantana	**0**	**0**	**3.17**	**2**	**3.5**	**3**
Lantana camara	**0**	**0**	**2.5**	**2.75**	**2.25**	**0**
Lantana montevidensis	**0**	**0**	**2.25**	**1**	**2**	**0**
Larrea tridentata	**0**	**5.69**	**3.29**	**0**	**0**	**3.5**
Lepidium lasiocarpum	**1**	**1.16**	**1**	**0**	**0**	**1**
Leucaena leucocephala	**0**	**0**	**3.5**	**0**	**0**	**4**
Leucophyllum	**0**	**0**	**2.12**	**0**	**1.33**	**1**
Leucophyllum candidum	**0**	**0**	**1.5**	**1**	**2**	**0**
Leucophyllum frutescens	**0**	**0**	**3.33**	**2**	**1.75**	**3.33**
Leucophyllum laevigatum	**0**	**0**	**5**	**5**	**0**	**0**
Leucophyllum pruinosum	**0**	**0**	**1**	**0**	**1**	**0**
Liliaceae	**0**	**0**	**3.25**	**3**	**4**	**0**
Lolium perenne	**1**	**0**	**1**	**1**	**1**	**0**
Lophocereus schottii	**0**	**0**	**1**	**0**	**1**	**0**

Lophocereus schottii fo *monstrosus*	**0**	**0**	**1.5**	**0**	**1.5**	**0**
Lycianthes rantonnetii	**0**	**0**	**1.5**	**2**	**0**	**1**
Lycium	**0**	**3.15**	**1**	**0**	**0**	**1**
Lysiloma watsonii	**0**	**0**	**2.25**	**1.33**	**2**	**3**
Malephora lutea	**0**	**0**	**1**	**0**	**0**	**1**
Malva parviflora	**1**	**1**	**1**	**1**	**1**	**1**
Malvaceae	**0**	**0**	**1**	**0**	**1**	**0**
Mammillaria	**0**	**2.4**	**2**	**1**	**3**	**0**
Marginatocereus marginatus	**0**	**0**	**1.4**	**0**	**1.25**	**2**
Medicago	**1**	**1**	**1**	**1**	**1**	**0**
Medicago lupulina	**2**	**0**	**1.27**	**1**	**1.25**	**0**
Melia azedarach	**0**	**0**	**3**	**4**	**2**	**0**
Melilotus	**0**	**0**	**1**	**0**	**1**	**0**
Mentha	**0**	**0**	**1**	**1**	**0**	**0**
Monolepis nuttalliana	**1**	**1.62**	**1.33**	**0**	**0**	**2**
Morus alba	**0**	**0**	**3.27**	**3.8**	**4.33**	**0**
Muhlenbergia	**0**	**0**	**1.5**	**1**	**2**	**0**
Muhlenbergia rigens	**0**	**0**	**3**	**0**	**1**	**0**
Myrtus communis	**0**	**0**	**3.75**	**5**	**4.5**	**0**
Myrtus communis Boetica	**0**	**0**	**1**	**1**	**0**	**0**
Nama hispida	**0**	**0**	**1**	**0**	**1**	**1**
Nandina domestica	**0**	**0**	**1.25**	**1.33**	**1**	**0**
Nerium oleander	**0**	**0**	**3.93**	**3.33**	**4.8**	**3.33**
Nicotiana obtusifolia	**0**	**0**	**1**	**0**	**0**	**1**
Nolina	**0**	**0**	**1**	**1**	**1**	**0**
Nolina microcarpa	**0**	**0**	**5**	**5**	**0**	**0**
Nolina nelsonii	**0**	**0**	**1**	**1**	**0**	**0**
Ocimum basilicum	**0**	**0**	**1**	**1**	**0**	**0**
Olea europaea	**0**	**0**	**2.1**	**2**	**1.5**	**5**
Oligomeris linifolia	**0**	**2**	**1**	**0**	**0**	**1**
Olneya tesota	**0**	**3**	**1.33**	**0**	**0**	**1**
Oncosiphon piluliferum	**1**	**1**	**1.2**	**1**	**1**	**1**
Opuntia	**0**	**2.5**	**2.36**	**1**	**1.25**	**3**
Opuntia articulata	**0**	**0**	**2**	**0**	**2**	**0**
Opuntia basilaris	**0**	**0**	**2**	**0**	**2**	**2**
Opuntia engelmannii var *linguiformis*	**0**	**0**	**1**	**0**	**1**	**1**
Opuntia microdasys	**0**	**0**	**1.5**	**0**	**1.5**	**0**
Opuntia phaeacantha	**0**	**0**	**4**	**0**	**4**	**0**
Opuntia santa-rita	**0**	**0**	**1.25**	**0**	**1.33**	**1**

Origanum vulgare	**0**	**0**	**1**	**0**	**1**	**0**
Oxalis	**0**	**0**	**1**	**1**	**1**	**0**
Oxalis corniculata	**0**	**0**	**1**	**1**	**1**	**0**
Pachycereus	**0**	**0**	**1.5**	**0**	**3**	**0**
Pachycereus pectin-aboriginum	**0**	**0**	**1**	**0**	**1**	**0**
Pachycereus pringlei	**0**	**0**	**1**	**0**	**1**	**0**
Pandorea jasminoides	**0**	**0**	**1**	**1**	**1**	**0**
Parkinsonia	**0**	**0**	**1.8**	**0**	**3**	**0**
Parkinsonia aculeata	**0**	**0**	**6.17**	**0**	**1.5**	**8.5**
Parkinsonia florida	**0**	**1.75**	**3.75**	**4**	**1.67**	**6.5**
Parkinsonia microphylla	**0**	**6.45**	**1.75**	**0**	**1**	**1.5**
Parkinsonia praecox	**0**	**0**	**1.4**	**1**	**0**	**1.67**
Parthenocissus quinquefolia	**0**	**0**	**1**	**0**	**1**	**0**
Passiflora	**0**	**0**	**1**	**1**	**0**	**0**
Pectocarya	**0**	**1**	**1**	**0**	**1**	**0**
Pectocarya recurvata	**1**	**1.09**	**1**	**0**	**1**	**0**
Peniocereus	**0**	**0**	**1**	**0**	**0**	**1**
Peniocereus greggii	**0**	**0**	**1**	**0**	**1**	**0**
Persicaria lapathifolia	**0**	**0**	**1**	**0**	**1**	**0**
Petroselinum	**0**	**0**	**1**	**0**	**1**	**0**
Petroselinum crispum	**0**	**0**	**1**	**1**	**0**	**0**
Phalaris minor	**1**	**0**	**1**	**1**	**1**	**1**
Philodendron	**0**	**0**	**1**	**1**	**0**	**0**
Phlox	**0**	**0**	**1**	**0**	**1**	**0**
Phoenix	**0**	**0**	**1**	**1**	**0**	**0**
Phoenix canariensis	**0**	**0**	**2.5**	**1**	**0**	**0**
Phoenix dactylifera	**0**	**0**	**3**	**0**	**3**	**0**
Phoenix roebelenii	**0**	**0**	**2.67**	**2**	**8**	**1**
Picea	**0**	**0**	**2**	**0**	**2**	**0**
Pinus	**0**	**0**	**2**	**1**	**0**	**4**
Pinus canariensis	**0**	**0**	**2**	**1**	**0**	**0**
Pinus eldarica	**0**	**0**	**5**	**0**	**0**	**5**
Pinus halepensis	**0**	**0**	**2.27**	**3.5**	**2**	**2.5**
Pistacia chinensis	**0**	**0**	**1**	**0**	**1**	**0**
Pistacia lentiscus	**0**	**0**	**1**	**0**	**0**	**1**
Pittosporum tobira	**0**	**0**	**1**	**1**	**1**	**0**
Platycladus orientalis	**0**	**0**	**1.75**	**2**	**0**	**1**
Plumbago auriculata	**0**	**0**	**1**	**0**	**1**	**0**
Poa annua	**0**	**1**	**1.15**	**1.33**	**0**	**1**
Poaceae	**1.33**	**1.57**	**1.59**	**1.75**	**1.6**	**1.25**
Polygonum	**1**	**0**	**1.67**	**1**	**0**	**0**

Polygonum argyrocoleon	**0**	**0**	**1.33**	**0**	**0**	**1**
Polygonum aviculare	**0**	**0**	**1.67**	**2**	**0**	**1**
Populus fremontii	**0**	**0**	**2.75**	**1.5**	**0**	**7**
Portulaca oleracea	**1**	**0**	**1**	**1**	**1**	**0**
Portulacaria afra	**0**	**0**	**2**	**2**	**2.5**	**0**
Prosopis	**1**	**1.71**	**2.89**	**2.75**	**2**	**4.2**
Prosopis chilensis	**0**	**0**	**1.6**	**0**	**1**	**1**
Prosopis glandulosa	**0**	**0**	**1**	**0**	**0**	**1**
Prosopis juliflora	**0**	**1**	**1.5**	**0**	**2**	**0**
Prosopis velutina	**0**	**2**	**1.67**	**0**	**1**	**3**
Prunus	**0**	**0**	**2**	**0**	**2**	**0**
Prunus persica	**0**	**0**	**3.5**	**0**	**6**	**0**
Punica granatum	**0**	**0**	**1**	**1**	**0**	**0**
Pyracantha	**0**	**0**	**1.5**	**0**	**2**	**0**
Pyracantha koizumii	**0**	**0**	**1**	**0**	**1**	**0**
Pyrus	**0**	**0**	**2**	**0**	**2**	**0**
Pyrus calleryana var *calleryana*	**0**	**0**	**2**	**1**	**0**	**3**
Quercus virginiana	**0**	**0**	**2.33**	**0**	**3**	**0**
Racosperma redolens	**0**	**0**	**1**	**0**	**1**	**0**
Rhaphiolepis	**0**	**0**	**1.33**	**0**	**2**	**0**
Rhus lancea	**0**	**0**	**2.57**	**2.25**	**3.75**	**3**
Ricinus communis	**0**	**0**	**2.5**	**3**	**0**	**2**
Rosa	**0**	**0**	**3.17**	**4**	**2.67**	**2**
Rosa banksiae	**0**	**0**	**1.5**	**1.5**	**2**	**0**
Rosmarinus officinalis	**0**	**0**	**1.67**	**1.33**	**2**	**2**
Ruellia peninsularis	**0**	**0**	**2.4**	**3**	**0**	**0**
Ruellia simplex	**0**	**0**	**2.71**	**2.75**	**2.67**	**2.25**
Rumex crispus	**0**	**0**	**1**	**0**	**1**	**0**
Rumex dentatus	**0**	**0**	**1**	**0**	**1**	**0**
Salix babylonica	**0**	**0**	**1**	**0**	**0**	**1**
Salsola tragus	**1**	**0**	**1**	**1**	**1**	**1**
Salvia greggii	**0**	**0**	**1**	**0**	**1**	**0**
Sambucus nigra	**0**	**0**	**1**	**0**	**1**	**0**
Sarcostemma cynanchoides	**0**	**1**	**1.4**	**0**	**1.5**	**1**
Schinus molle	**0**	**0**	**2**	**2**	**0**	**0**
Schinus terebinthifolius	**0**	**0**	**1.4**	**1**	**1**	**1**
Schismus	**1**	**1**	**1.2**	**1**	**0**	**0**
Schismus arabicus	**1.5**	**1**	**1**	**1**	**1**	**1**
Schismus barbatus	**1**	**1.31**	**1.08**	**1**	**1.33**	**1**
Senna artemisioides	**0**	**0**	**2.2**	**0**	**2**	**2.33**

Senna artemisioides ssp *filifolia* Randall	**0**	**0**	**1**	**1**	**1**	**1**
Senna covesii	**0**	**3**	**2**	**0**	**0**	**3**
Senna phyllodinea	**0**	**0**	**3**	**0**	**0**	**4**
Simmondsia chinensis	**0**	**4.7**	**5**	**0**	**5**	**0**
Sisymbrium irio	**1.22**	**1.38**	**1.22**	**1.1**	**1.11**	**1.17**
Solanum elaeagnifolium	**1**	**0**	**1**	**1**	**1**	**1**
Solanum lycopersicum	**0**	**0**	**1.33**	**2**	**1**	**1**
Sonchus	**1.33**	**1**	**1.37**	**1.25**	**1.14**	**1.4**
Sonchus asper	**1**	**0**	**1.12**	**1**	**1.5**	**0**
Sonchus oleraceus	**1**	**1**	**1.2**	**1.33**	**1.33**	**1.33**
Sphaeralcea	**0**	**1**	**1.33**	**2**	**0**	**0**
Sphaeralcea ambigua	**5**	**3.75**	**2.2**	**5**	**1**	**0**
Sphaeralcea emoryi	**0**	**5**	**1**	**1**	**0**	**1**
Stellaria media	**0**	**1**	**1**	**1**	**1**	**0**
Stenocereus	**0**	**0**	**1**	**0**	**1**	**0**
Stenocereus thurberi	**0**	**0**	**1**	**0**	**0**	**1**
Stephanomeria pauciflora	**1**	**1.38**	**1.23**	**0**	**1**	**1**
Strelitzia nicolai	**0**	**0**	**3**	**3**	**0**	**0**
Syagrus romanzoffiana	**0**	**0**	**2.67**	**0**	**5**	**3**
Tamarix chinensis	**0**	**0**	**1.25**	**1**	**0**	**2**
Taraxacum	**0**	**0**	**1**	**0**	**0**	**1**
Taraxacum officinale	**0**	**0**	**1**	**1**	**1**	**0**
Tecoma capensis	**0**	**0**	**1.9**	**1.75**	**1.5**	**0**
Tecoma stans	**0**	**0**	**3.1**	**2.67**	**3**	**0**
Thevetia peruviana	**0**	**0**	**1.29**	**1**	**1.5**	**1**
Thymophylla pentachaeta	**0**	**0**	**1.11**	**1**	**1**	**1.33**
Tipuana tipu	**0**	**0**	**1.67**	**0**	**0**	**2**
Torilis nodosa	**0**	**0**	**1**	**1**	**0**	**0**
Trachelospermum jasminoides	**0**	**0**	**2**	**2**	**0**	**0**
Tradescantia pallida	**0**	**0**	**2**	**0**	**0**	**2**
Tribulus terrestris	**0**	**0**	**1.12**	**1**	**2**	**1**
Ulmus parvifolia	**0**	**0**	**1.86**	**1.88**	**1.75**	**1.25**
Vachellia farnesiana	**0**	**0**	**1**	**0**	**1**	**0**
Verbesina encelioides	**1**	**0**	**1**	**0**	**1**	**1**
Veronica arvensis	**0**	**0**	**1**	**1**	**0**	**0**
Vigna caracalla	**0**	**0**	**1**	**0**	**1**	**0**
Vinca	**0**	**0**	**1**	**0**	**1**	**0**
Vitex agnus castus	**0**	**0**	**1**	**1**	**0**	**0**
Vulpia	**0**	**0**	**1**	**1**	**0**	**0**
Vulpia octoflora	**0**	**1.11**	**1**	**1**	**0**	**0**

Washingtonia	**0**	**0**	**2.71**	**0**	**3**	**0**
Washingtonia filifera	**1**	**0**	**1.2**	**1**	**1**	**1**
Washingtonia robusta	**0**	**0**	**2.22**	**1**	**5**	**1**
Yucca	**0**	**5**	**1.78**	**1**	**2**	**1**
Yucca aloifolia	**0**	**0**	**1.5**	**1**	**2**	**0**
Yucca baccata	**0**	**0**	**1**	**0**	**0**	**1**
Yucca brevifolia	**0**	**0**	**1.5**	**0**	**0**	**1**
Yucca elata	**0**	**0**	**1**	**0**	**1**	**1**
Yucca gloriosa var *recurvifolia*	**0**	**0**	**1.33**	**0**	**2**	**0**
Ziziphus obtusifolia	**0**	**4**	**1**	**0**	**0**	**1**

APPENDIX B

FIGURE S2.1. CASE-SHILLER HOME PRICE INDEX FOR PHOENIX, AZ.

ACCOMPANIES CHAPTER 2.

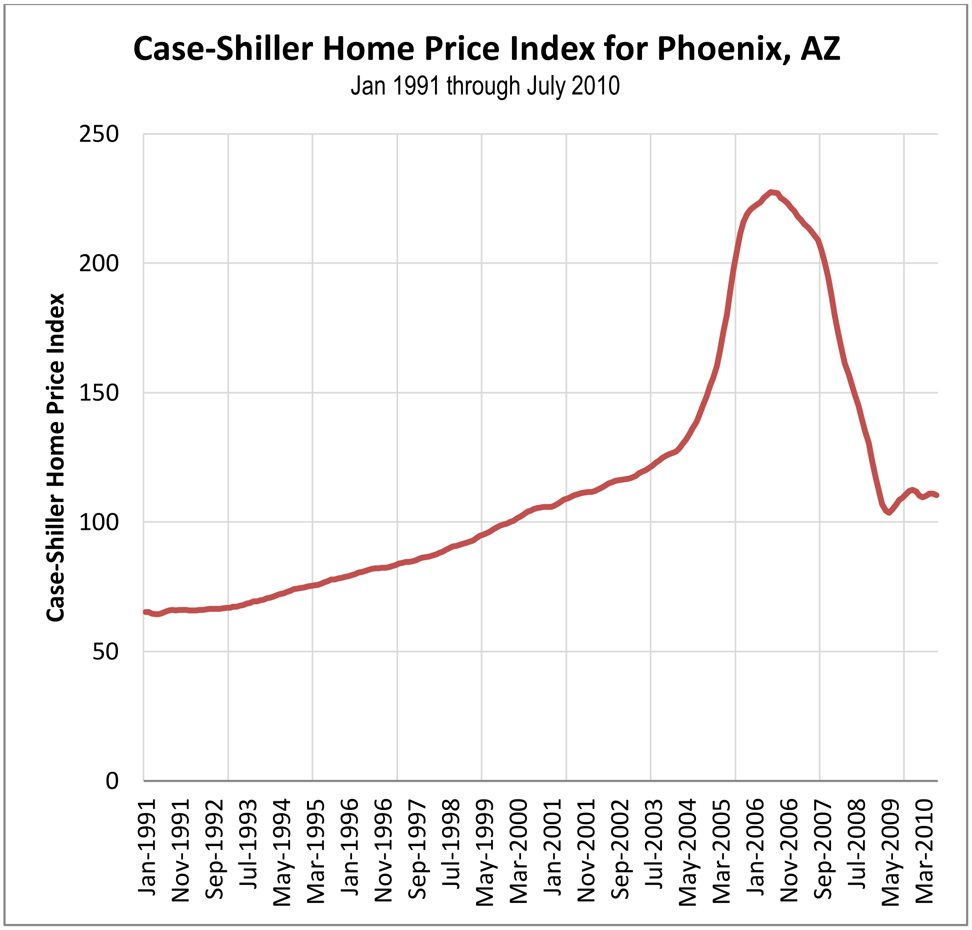

Figure S2.1. Trend in Phoenix home value using the Case-Shiller home price index for January 1991 through July 2010. Data exported from S&P Dow Jones Indices (https://us.spindices.com/indices/real-estate/sp-case-shiller-az-phoenix-home-price-index).

APPENDIX C

FIGURE S2.2. SEASONAL PRECIPITATION TREND FOR PHOENIX, AZ.

ACCOMPANIES CHAPTER 2.

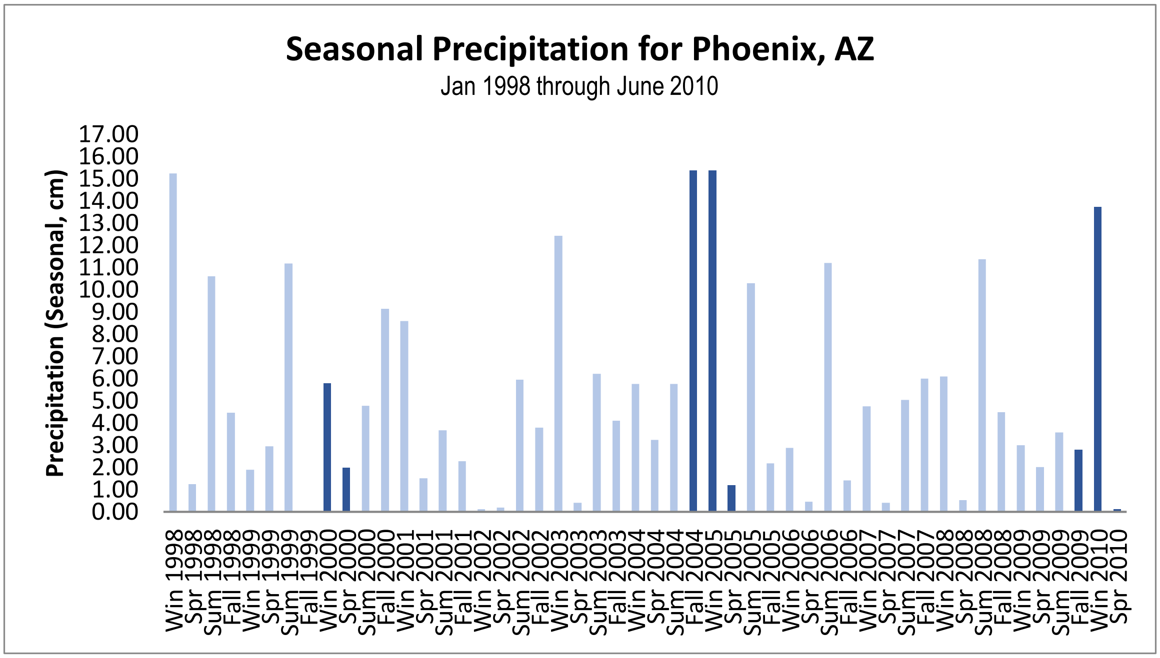

Figure S2.2. Variation in seasonal (3-month sum) precipitation for Phoenix, AZ. Graph shows January 1998 through June 2010. Seasonal precipitation calculated using monthly precipitation values of three Phoenix-area Arizona Meteorological Network (AZMET) weather stations (http://ag.arizona.edu/azmet/azdata.htm).