


Boom–bust economics and vegetation dynamics in a desert city: How strong is the link?

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Citation: Ripplinger, J., S. L. Collins, A. M. York, and J. Franklin. 2017. Boom–bust economics and vegetation dynamics in a desert city: How strong is the link? *Ecosphere* 8(5):e01826. 10.1002/ecs2.1826

Abstract. In cities, human activities such as supplemental watering and plantings of ornamental species are thought to decouple vegetation diversity from biophysical processes. Consequently, socioeconomics may arguably be one of the most important factors influencing the pattern and dynamics of vegetation in urban ecosystems. Socioeconomic disturbances, like The Great Recession of 2007 to post-2010, disrupt normal social and economic activity causing changes to the ecology of cities that have yet to be examined. Using the rapidly growing metropolitan area of 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 changing plant community composition and increasing richness with socioeconomics of the housing boom, (from 2000 to 2006) followed by the Great Recession (from 2007 to post-2010). Specifically, we found that the housing market boom–bust episode acted as a socioeconomic driver of overall plant community composition. However, plant species with annual reproduction strategies were instead linked to the previous winter's precipitation and did not respond to socioeconomic disturbance, a behavior expected in native desert plant communities, but not from constructed and managed urban plant communities. This study demonstrates how disturbances to socioeconomic components of ecosystems can result in changes to plant community diversity and dynamics; though, biophysical drivers remain important to short-lived annual species regardless. Undertakings that aim to maintain or increase urban biodiversity for ecosystem services and 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.

Key words: biodiversity; cities; disturbance; economic disruption; plant communities; precipitation; recession; social–ecological systems; urbanization.

Received 5 April 2017; accepted 10 April 2017. Corresponding Editor: Debra P. C. Peters.

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INTRODUCTION

As the global human population becomes ever more urbanized, we are increasingly exposed to designed and built infrastructure, whether “gray” infrastructure like roads and buildings or “green” infrastructure like parks and yards (Childers et al. 2015). The result of this expanding infrastructure

may be urban ecosystems that are decoupled from fundamental biophysical processes (Shochat et al. 2006). Like their non-urban counterparts, urban ecosystems are highly dynamic (Ripplinger et al. 2016), and both cultivated and naturalized species distributions vary widely among cities (La Sorte et al. 2014, Jenerette et al. 2016). Urban ecosystems can be extremely biodiverse (Jenerette

et al. 2016), and landscapes like yards, gardens, and parks offer residents (sometimes their only) opportunity to enjoy nature. When people have positive experiences in nature, they are likely to be physically (Maas et al. 2006) and mentally healthier (Mayer and Frantz 2004, Fuller et al. 2007), as well as to engage in sustainable behaviors that improve ecosystem health (Tzoulas et al. 2007). Researchers are beginning to understand preferences and decisions around residential vegetation from a socioecological perspective (Larsen and Harlan 2006, Larson et al. 2009, 2017), and to apply the concept of ecological disturbance to urban systems (Pickett and White 1985, Grimm et al. 2017). Yet to date, there has been little work on how socioeconomic disturbance might affect urban vegetation dynamics. Understanding urban residential diversity and drivers will be imperative to designing resilient and sustainable urban ecosystems for human and ecosystem health outcomes.

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 tree canopy cover to the capacity of a household to manage vegetation (Grove et al. 2014). Unevenness in socioeconomic capacity across social strata, for example, may result in varied preferences, for example, mature trees, established lawns, or perennial gardens (Grove et al. 2006, 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 as socioeconomic status increases, people choose to occupy or install landscapes that incidentally have higher diversity. Such studies account for the underlying financial capacity of a household to regulate vegetation cover and diversity in their yards.

Pioneer plant species rely on disturbance for establishment, have broad physiological tolerances, are able to acclimate to a wide range of conditions, and have rapid growth rates (e.g., weedy annual plants; Rejmanek and Richardson 1996). In urban systems, when people stop maintaining

landscapes, weedy species are presented with an opportunity to increase in abundance both locally and spatially. For our purposes, the term “weedy” includes plants sometimes referred to as “volunteers” that appear spontaneously in the landscape rather than having been intentionally planted there. Volunteers may be native or non-native annual or perennial plants, or may even be reverse invaders—native plant species that have become weedy plants within city boundaries. Moreover, plants used in urban settings are often non-native, and though most are not invasive, some escape from agricultural or horticultural uses to become nuisances in urban and non-urban landscapes (Reichard and White 2001). Additionally, successful invaders often go through multiple introduction events (Gray 1986), and adjacent landscapes under economic stress may provide a pool of colonizing species. As a result, colonization pressure from annual and introduced species increases on landscapes with adjacent unmaintained landscapes.

Biophysical disturbances to urban ecosystems, such as floods, fire, and pests, are well studied and have been shown to alter soil resources (Pickett and Cadenasso 2009), erode urban stream beds (Walsh et al. 2005), and increase human–wildlife conflict and destruction of homes by wildfires (Radeloff et al. 2005). For many such disturbances, viable engineered solutions exist. For instance, cities in the U.S. Southwest have had success in flood management through the installation of constructed retention basins and other forms of designer landscapes (Palmer et al. 2004, Grimm et al. 2005, Roach et al. 2008). Engineered solutions mitigate many deleterious effects of biophysical disturbances in urban systems, but overlook a potentially important driver of urban ecosystem dynamics—socioeconomic disturbance. Anthropogenic disturbance is often defined as a relatively abrupt, physical impact that is detrimental to the biophysical environment (e.g., forest clear-cutting). We define socioeconomic disturbance, by contrast, as a disruption to economic and social activity causing a measurable change to a social–ecological system (*sensu* Pickett et al. 1989). The example of socioeconomic disturbance we explore is the change in urban vegetation diversity in response to a boom–bust episode in the housing market. We focus here on ecological response to economic and biophysical drivers, but propose

that socioeconomic disturbances can be conceptualized similarly regardless of whether the variables of interest are social, economic, or biophysical.

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. Central Arizona-Phoenix Long-Term Ecological Research (CAP LTER) program conducted a comprehensive survey of urban vegetation composition in 2000, 2005, and 2010, providing a unique opportunity to examine changing residential vegetation composition in response to extensive mortgage foreclosures and other effects of the Great Recession. Species distributions in cities are markedly influenced by human decisions (Jenerette et al. 2016), particularly in highly managed areas like residential neighborhoods (Knapp et al. 2012). In residential areas, there is a direct connection between the “land manager” (e.g., a household or property manager) and the landscaping, where plants in the residential landscape are there because land managers do or do not want them there and landscaping diversity is at the behest of the manager. However, climate factors have been shown to broadly regulate urban plant communities (La Sorte et al. 2014, Avolio et al. 2015), and desert ecosystems are particularly sensitive to precipitation variation. Because desert ecosystems are water limited, supplemental watering and/or precipitation directly regulate plant productivity and diversity. These competing drivers have led to multiple hypotheses of urban plant biodiversity dynamics.

Our overall objective was to examine the effect of socioeconomic disturbance factors vs. biophysical variables on changes in urban plant species diversity and composition. We expected socioeconomic disturbance to have a stronger influence than biophysical variables because urban vegetation has been shown to be structured by household-level decisions more than by climate or edaphic factors (Seto and Kaufmann 2003, Cook et al. 2012). We hypothesized that vegetation diversity would be negatively affected by the Great Recession. This outcome would be the result of a reduction in non-adapted introduced plants, for example, from reduced supplemental watering. Socioeconomic distress would limit supplemental watering and planting of ornamentals, reducing residential plant diversity. Alternatively,

vegetation diversity might increase with socioeconomic distress. This outcome would be the result of expansion of fast-growing, desert-adapted plants, for example, due to neglect of weed abatement activities. The Great Recession would release residential 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. This result would increase plant community diversity and shift composition from pre- to post-recession. We asked the following questions: What is the strength of socioeconomic disturbance (e.g., value of foreclosed housing, foreclosure density) vs. biophysical drivers (e.g., precipitation) in predicting residential vegetation diversity and composition? How do residential vegetation diversity and composition 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, Arizona, USA, along with several smaller adjacent cities, and is home to more than 4 million people. The CAP LTER study area is 6400 km² and encompasses the Phoenix Metropolitan Area as well as surrounding agriculture land and the Sonoran Desert ecoregion (Fig. 1). The CAP LTER “Survey 200” used a systematic random design to distribute ~200 survey plots throughout urban Phoenix as well as surrounding desert and agricultural areas (Hope et al. 2003). Urban survey plots occurred in a range of locations, from residential yards to shopping centers to transportation corridors. From these, we selected plots that were classified as residential—roughly 20% of the 200 random plots, enabling a total of 39, 45, and 35 residential surveys from the three survey periods, respectively, to be included in this study. Plots were classified as residential if >50% of the plot was designated as a residential-type land use, for example, single-family yard or a property with multi-housing units. Surveys were conducted at the same plots from year to year, though land-use change and restricted access resulted in uneven numbers of residential plots surveyed across years. Each plot consisted of a single 30 × 30 m square where vegetation was re-surveyed every

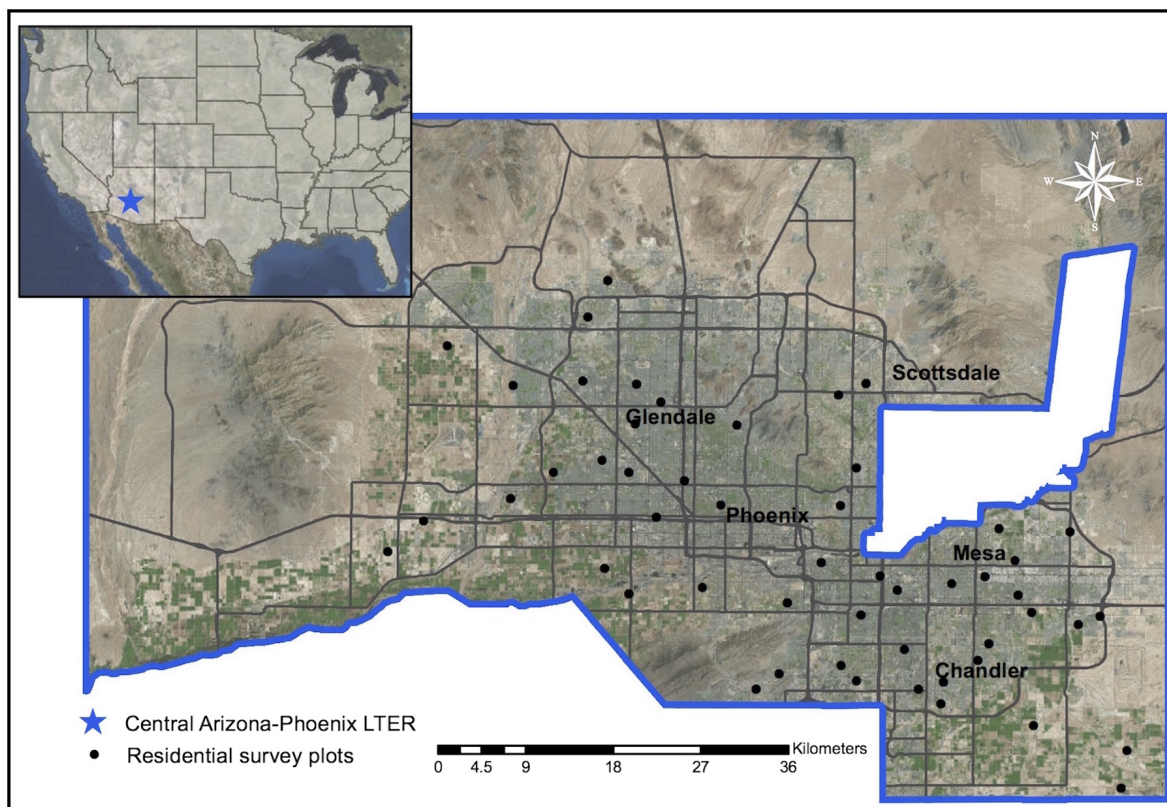


Fig. 1. Map of CAP LTER study area. Solid circles indicate locations of residential vegetation surveys. Blue star in inset map (upper left) represents location of CAP LTER in Arizona, USA.

five years (2000, 2005, 2010). Plots were randomly placed a minimum of 500 m from the nearest adjacent plot to reduce spatial autocorrelation. At each plot, all vascular plant species were identified and the number of individuals was counted. Species were assigned to the following growth form categories: native/non-native, herbaceous annuals, cacti/succulent, 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. Our goal was to assess the relative importance of dynamic socioeconomic disturbance and biophysical factors, so we considered 17 predictors of the diversity and composition of plant species (see Appendix S1: Table S1). Selected variables were hypothesized to be factors affecting urban and residential

patterns of vegetation diversity, for example, home value, socioeconomic status (Hope et al. 2003, Martin et al. 2004, Kinzig et al. 2005), legacy effects of land use (Cook et al. 2012, Johnson et al. 2015), home construction year (Larson et al. 2017), foreclosure frequency, as well as fundamental climate and biophysical factors, for example, precipitation, soil, minimum temperature (Grace 1999). Socioeconomic predictors specific to the housing market boom–bust were taken from publicly recorded real estate data compiled by The Information Market (www.theinformationmarket.com). Home values and other housing variables were obtained 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 generalized least-squares (GLS) regression models, ensuring that all model assumptions were met. Because surveys were repeated on the same plots over three time periods and expected to correlate from one survey year to the next, we included an autoregressive process of order 1 (AR[1]) to account for temporal autocorrelation among plots. Species richness across all plots and survey periods was modeled as a function of several hypothesized predictor variables (Table 1). We ran separate GLSs for species richness of (1) all plant species, (2) annual species only, (3) introduced species only, using the “nlme” package version 3.1-128 in R (R Development Core Team 2015, Pinheiro et al. 2017).

Analyses of change in plant community composition were based on abundance data for all plant species. The constrained ordination technique, canonical correspondence analysis (CCA; Ter Braak 1986), was used to relate variation in plant species composition to socioeconomic and biophysical variations in the environment. Hierarchical cluster analysis can be used to identify compositionally similar groupings of plots. Groups are identified by calculating ecological distance between every possible pair of clusters and merging similar pairs until an ecologically meaningful number of clusters have been reached. We chose to use Euclidean distance which was the most effective when compared to other distance measures in obtaining separation among groups of clusters, and Ward’s method to

minimize between-group variance. Ward’s method of hierarchical agglomerative clustering progressively merges clusters to minimize within-group variance by minimizing sums of squared error (Ward 1963). Based on species composition, we identified distinct groups of plots for each survey year separately. Changes in the dominant species among groups defined by clustering over time and cluster membership of plots between survey years were identified. Dominant species were defined as the ~ two species with the highest count of individuals in each group defined by clustering (Fig. 2). Compositional analyses were performed using the packages “vegan” version 2.4-1 and “labdsv” version 1.8-0 in R (R Development Core Team 2015, Oksanen et al. 2016, Roberts 2016).

To examine trends in diversity from 2000 to 2010, species richness in each survey year was calculated for plots in three ways: (1) for all plant species, (2) for annual species only, and (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. To assess potential sampling bias, species accumulation curves were calculated; the curves showed saturating behavior, illustrating that the survey plots adequately represent the potential species richness in the landscape (Appendix S1: Fig. S1). Analyses of species diversity were also conducted using Shannon index, but direction of results were comparable so we report species richness results.

Table 1. Predictor variables used in best-fit generalized least-squares (GLS) regression of plant species richness; model selected using Akaike’s Information Criterion (AIC).

| Variables | Predictors | Responses | | |
|---------------|-------------------------------|---------------------|---------------------|---------------------|
| | | Total richness | Introduced richness | Annals richness |
| Biophysical | Precipitation, winter | −0.01627 (0.02627) | −0.01108 (0.01801) | 0.01416** (0.00806) |
| | Soil type | 0.08332 (0.07679) | 0.05880 (0.05309) | 0.03114 (0.02377) |
| | Survey year (residuals) | 0.98498* (0.05390) | 0.98277* (0.05940) | 1.01986* (0.02531) |
| Socioeconomic | Assessed value at foreclosure | 0.00001 (0.00001) | 0.00 (0.00) | 0.00 (0.00) |
| | Foreclosure density | 0.02120** (0.01159) | 0.01479** (0.00801) | 0.00450 (0.00338) |
| | Time since foreclosure | 0.00318 (0.00682) | 0.00216 (0.00470) | 0.00154 (0.00207) |
| | Time since land-use change | 0.04350* (0.02086) | 0.03021* (0.01427) | 0.00776 (0.00569) |
| | Year built | 0.07840* (0.03456) | 0.05373* (0.02381) | 0.01272 (0.00981) |

Notes: Estimated coefficients shown, with standard error given in parentheses. Level of significance denoted by asterisks (* $P \leq 0.05$, ** $P \leq 0.08$).

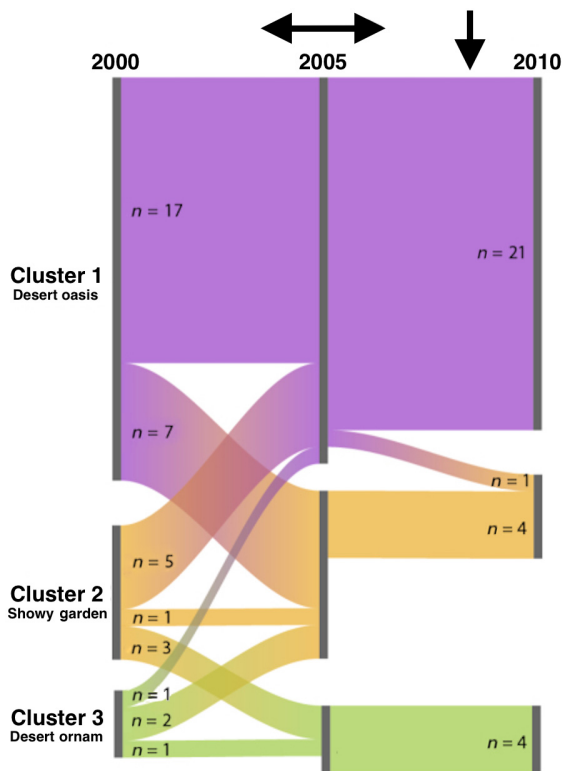


Fig. 2. Graphical representation of land-use cluster transitions from 2000 (left) to 2010 (right) between plant associations as defined by hierarchical cluster analysis. The flows (left to right) indicate direction of plot transitions among plant associations over time. Number of plots transitioning among clusters across survey years is represented by width of flows and inset n -values. Approximate window of housing market bubble is denoted by black horizontal arrow at top-center, and bottom of housing market crash is denoted by black vertical arrow at top-right. Uneven plot numbers across years (39 in 2000, 45 in 2005, and 35 in 2010) resulted from access limitations.

RESULTS

Biophysical and socioeconomic drivers of residential vegetation diversity

After excluding highly correlated variables (Pearson's correlation ≥ 0.6), the following predictors were included in regression models: assessed value at foreclosure, density of foreclosures nearby, soil type, time since foreclosure, winter precipitation, year home was built, and time since plot changed to residential land use (Table 1).

For total plant species richness, we compared models with and without temporal autocorrelation structures and found considerable improvement in model explanatory power with the inclusion of a first-order autoregressive process (AR[1]) in the model and partialing to control for survey year (lower Akaike's Information Criterion [AIC] = $\Delta 96$). Foreclosure density, time since land-use change, and year built had a significant positive effect on total species richness (Table 1). For introduced species (lower AIC = $\Delta 80$), foreclosure density, time since land-use change, and year built had a significant positive effect on total species richness, while winter precipitation had a non-significant negative effect (Table 1). For annual species richness, a model with temporal autocorrelation was again selected (lower AIC = $\Delta 253$) and winter precipitation was the only significant explanatory variable (Table 1).

Impact of biophysical and socioeconomic factors on community composition

Plot species composition differed significantly from year to year ($P < 0.01$). Inter-annual compositional variation was structured by home value, winter precipitation, density of nearby foreclosures, time since foreclosure, and year built (Fig. 3). Plots were arrayed along the x -axis of the CCA plot by the negatively correlated

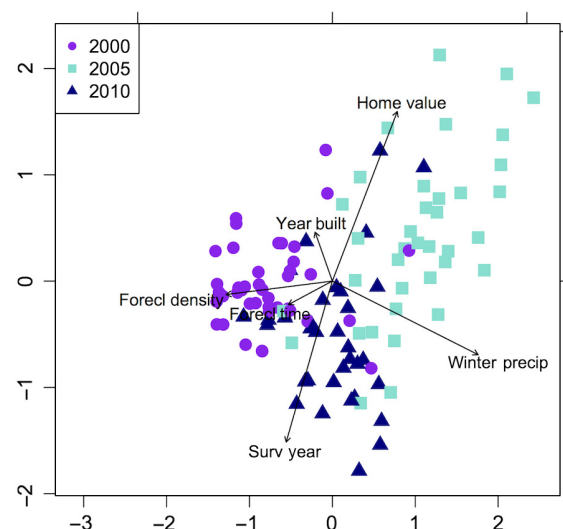


Fig. 3. Canonical correspondence analysis of residential survey plots as structured by biophysical and socioeconomic disturbance variables.

variables of increasing winter precipitation and decreasing foreclosure density and time since foreclosure. Along the y -axis, plots were arrayed by the negatively correlated variables of increasing value of home value and decreasing survey year. In 2000, community composition was driven by low winter precipitation and low foreclosures. Year 2005 plots were more often associated with high-value foreclosed homes, low density of nearby foreclosures, and high winter precipitation. The composition of plots in 2010 was strongly structured by lowest home values.

Changes in residential vegetation through the housing boom–bust cycle

Combined across all years, residential vegetation surveys included a total of 611 plant species (Appendix S1: Table S2). The total number of species found in each survey year (gamma diversity of the residential landscape) trended upward from 2000 to 2005 to 2010 (Fig. 4). Site-level annual species richness peaked in 2005, while total and introduced species richness increased each survey year (Fig. 4).

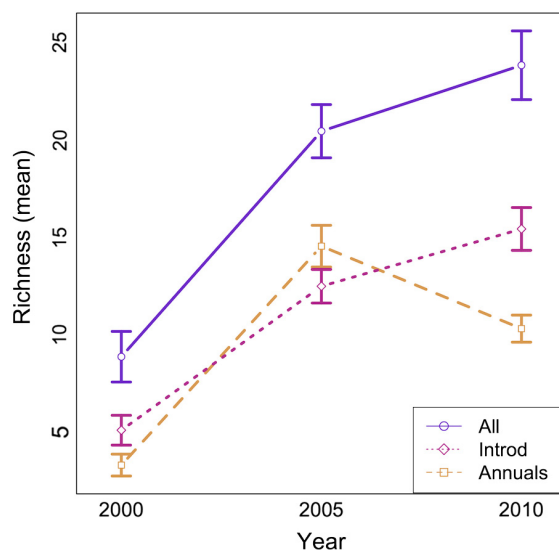


Fig. 4. Site-level plant species richness of growth forms by provenance and longevity. Points are mean species richness values of residential plots. Values given for periods representing pre-recession (2000), housing boom (2005), and post-recession (2010). Error bars show 95% confidence intervals.

Cluster analysis of plot composition in 2000 (PERMANOVA $R = 0.073$, $P = 0.357$, 999 permutations) resulted in a group of “desert oasis” plots, the largest group ($n = 27$) identified by clustering. This group of plots was dominated by *Washingtonia* spp., a widely planted ornamental palm (of a genus that is native to the region), and introduced ornamental fruit trees (*Citrus* spp.; Fig. 2). The “showy garden” group ($n = 8$) was dominated by common ornamental garden species, like roses (*Rosa* spp.) and lavender (*Lavandula* spp.). Finally, a third group of “desert ornamental” outlier plots ($n = 4$) was dominated by prickly pear cactus, *Opuntia* spp., a native genus that is also used in landscaping, and the introduced horticultural flower, *Crocus* spp.

Cluster analysis of plot composition in 2005 (PERMANOVA $R = 0.027$, $P = 0.153$, 999 permutations) generated a “desert oasis” group ($n = 31$) dominated by the small, native horticultural tree, *Acacia farnesiana*, and the weedy introduced herbaceous forb *Plantago* sp. (Fig. 2). In 2005, the “desert oasis” group incorporated five plots that had been members of the “showy garden” group in 2000 (Fig. 2). Also in 2005, the “showy garden” group incorporated seven plots that had been members of the “desert oasis” group in 2000. The “showy garden” plots ($n = 10$) were dominated by *Leucophyllum frutescens* and *Nerium oleander*, both introduced perennial species with horticultural value and long-lived flowers. This group primarily comprised plots that had been associated with the “desert oasis” and “desert ornamental” groups in 2000. Finally, a small group of “desert ornamental” outlier plots ($n = 4$) was dominated by the native but often weedy shrub *Isocoma acradenia*.

For cluster analysis of 2010 (PERMANOVA $R = 0.036$, $P = 0.126$, 999 permutations), most plots remained in their 2005 cluster groupings, but the dominant species that characterized these groups changed (Fig. 2). The “desert oasis” group ($n = 25$) was dominated by *Atriplex elegans*, a native but weedy saltbush shrub, and the introduced ornamental tree/shrub, *N. oleander*. The “showy garden” 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 “desert ornamental” cluster consisted of the same small group of plots ($n = 4$) as in

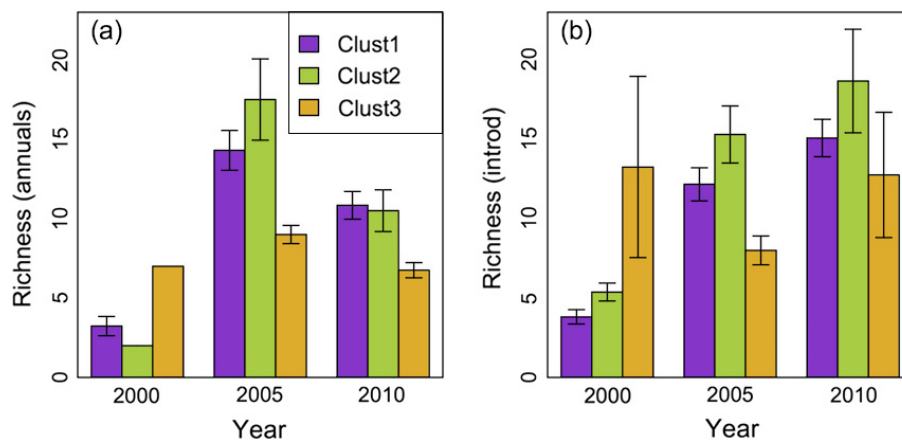


Fig. 5. Plant species richness of growth forms for plant associations defined by hierarchical clustering. Changes in plant species richness shown for (a) annual plant species only and (b) introduced species only. Values given for periods representing pre-recession (2000), housing boom (2005), and post-recession (2010). Confidence intervals (95%) shown by error bars.

2005, but dominants changed to the introduced ornamental shrub Natal Plum *Carissa macrocarpa*, and the native ornamental tree Palo Verde *Parkinsonia aculeata*.

To assess how richness of annual plants and introduced species changed through time, we plotted the distribution of site-level species richness for groups defined by clustering for each survey year (Fig. 5). Clusters 1 (desert oasis) and 2 (showy garden) included the majority of survey plots each year, with Cluster 3 (desert ornamental) consisting of fewer than five plots in each year. For annual species (Fig. 5a), all groups defined by clustering 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 group was relatively stable across survey years. For introduced species (Fig. 5b), 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, thus resulting in no significant trend over time.

DISCUSSION

We investigated the relative impact of the Great Recession boom–bust episode and biophysical drivers on urban residential plant communities. We found that neighborhood foreclosure rates

increased plant species richness, particularly for introduced species (Table 1), consistent with our predictions for socioeconomic mechanisms of residential vegetation dynamics. In contrast, annual plant diversity responded positively only to antecedent winter precipitation (Table 1), revealing that annual plants as a functional group are not decoupled from biophysical drivers. Foreclosures and home value played a strong role alongside winter precipitation in structuring overall composition of residential plant communities (Fig. 3). We also found that composition and species richness shifted over the course of the housing boom (pre-2000 to 2006) to housing bust cycle of the Great Recession (2007 to post-2010; Figs. 2–5).

Our findings help resolve issues with predictions of urban biodiversity under both socioeconomic and biophysical pressures. Here, both mechanisms act to structure urban plant communities, but differently among plant growth forms. Socioeconomics have been shown to increase urban vegetation diversity (Hope et al. 2003, Grove et al. 2014), and while our study supports those findings for total and introduced species richness, they also show that its influence is complex. For example, annual plant species were greatly influenced by precipitation, and legacy effects of land use and urbanization reflect the potential for other socioeconomic or cultural variables to influence patterns of urban biodiversity. Still, the contrasting responses among vegetation

growth forms to socioeconomic and biophysical drivers suggest that urban vegetation remains coupled in surprising ways to ecosystem processes in structuring plant distributions and dynamics.

Our study provides an example by which socioeconomic disturbance can affect urban plant diversity. However, rather than the Great Recession driving residential vegetation change, we found that the biggest vegetation change detected came with the housing bubble rather than the bust. 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), so future surveys might reveal new patterns of vegetation change following the boom–bust cycle. Notably, the housing bubble rather than the recession appeared to have a stronger effect on plant community diversity and composition. The foreclosure rate in 2010 was 39%, as contrasted with average foreclosure rate of ~5% in previous years (<https://asunow.asu.edu/content/final-phoenix-area-foreclosure-numbers-2011>), the result being a more apparent “luxury effect” (Hope et al. 2003) of the housing bubble, than the ecological release of pioneer species we predicted to occur when the bubble burst. Shifts in economic and cultural factors accompanying the housing bubble produced numerous land-use transitions, reflecting preferences for lush desert oasis and showy garden plants in landscape management activities (Fig. 2). In response to the housing boom, in 2005, a number of plots transitioned from the “desert oasis” aesthetic to the “showy garden” aesthetic and also from “showy garden” to “desert oasis.” The plots that transitioned to or remained in the “showy garden” cluster were generally newer homes (built ~1988–1990 vs. ~1974–1978) with higher home value (Δ \$106K higher) and larger home size (Δ 650 ft² larger) than plots in the “desert oasis” cluster in 2005. These results suggest an increasing disparity in landscaping vegetation, separating smaller, lower-value homes with less yard biodiversity from larger, higher-value homes with more yard biodiversity (Figs. 2 and 5).

During wet years in water-limited desert systems, increases in precipitation typically lead to increases in productivity. However, previous research has shown that managed vegetation can be decoupled from the influence of precipitation (Buyantuyev and Wu 2012). We found that

socioeconomic factors successfully predicted total and introduced plant species richness, but importantly, winter precipitation was an equally strong predictor of annual plant species richness (Table 1). This increase in richness was evident across all site types, regardless of cluster association (Fig. 5a). Rather than coupling with the housing boom–bust cycle, annual plant species were instead highly correlated with antecedent precipitation, much like behaviors of native desert annual communities. In urban settings, wet conditions that otherwise increase abundance of weeds likely lead to efforts to suppress them (e.g., *Baccharis sarothroides*, *Cynodon dactylon*, *Lactuca serriola*; Appendix S1: Figs. S2 and S4), promoted rapid reproduction of annual plants in response to the short-term pulse of precipitation. High seasonal precipitation that occurred in the months prior to the 2005 field survey (Appendix S1: Fig. S2) could promote an increase in vegetation productivity and diversity (Hall et al. 2011, Mulhouse et al. 2017). 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. 5a). 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 (Appendix S1: Fig. S3). The bubble provided additional economic resources for landscape plantings of ornamental annuals, but also provided opportunities for weedy annuals in neighborhoods cleared for development with individual properties in various stages of construction. Alternatively, an increase in annual weeds could have been due to lots being cleared for development during the housing boom. However, the diversity of introduced species did not increase over the same time period (Fig. 5b), nor did total plant species richness (Fig. 4). Instead, environmental and economic conditions resulted in richness gradually increasing from 2005 into 2010 for all species and for introduced species (Fig. 4), with the lowest total and introduced species richness occurring on plots classified as outliers for their species composition (Fig. 5b). This could have resulted from 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 (Fig. 4) and lower abundance of select weedy species (Fig. 3).

Faeth et al. (2011) suggested that households may prefer to include a wide variety of plants in their yards, regardless of provenance of the plant species, so underlying preferences may in fact tend toward greater diversity rather than native diversity. We found, however, that plant community composition during the housing boom in 2005 was most similar to vegetation found at affluent outlier clusters—clusters whose composition was dominated by ornamental plants native to the Sonoran Desert. One possible explanation for this finding is that personal preferences tend toward yards with more native ornamentals, with households able to afford native ornamentals in 2005, but not in 2000 and 2010 when boom-time economics were not in play. Future research into landscaping preferences among various neighborhoods and demographic groups would further clarify this finding (e.g., Uren et al. 2015).

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 socioeconomic driver of changes in plant community richness and composition. Additionally, in an arid urban ecosystem, precipitation remains important to annual species abundance, reinforcing the premise that urban vegetation diversity is partially coupled to biophysical drivers despite human constraints on biophysical processes of cities. Undertakings that aim to maintain or increase urban biodiversity for ecosystem services and human well-being need to systematically approach the effects of socioeconomic fluctuations on urban flora. A key direction in our continued efforts to build urban ecological theory and develop a broader understanding of urban vegetation dynamics will include testing these findings across globally distributed cross-site comparisons in older and newer, mesic and arid, and growing and shrinking cities.

ACKNOWLEDGMENTS

This material is based upon work supported by the National Science Foundation under Grant Number BCS-102686, Central Arizona-Phoenix Long-Term Ecological Research (CAP LTER) Project. CAP LTER vegetation data used in this study are available at <https://caplter.asu.edu/data/data-catalog/?id=278>. This research received support for foreclosure data from the National Science Foundation under Grant SES-0345945, Decision Center for a Desert City, and Grant SES-0951366, Decision Center for a Desert City II: Urban Climate Adaptation. Additional support was provided by the National Science Foundation through the Urban Sustainability Research Coordination Network (Grant No. 1140070). We thank Stevan Earl for sharing his expertise and insights about the CAP LTER data. Dan Childers and two anonymous reviewers provided useful comments that improved a previous draft of the manuscript. And finally, we are grateful to Joanna Merson for designing the Fig. 2 illustration.

LITERATURE CITED

- Avolio, M. L., D. E. Pataki, S. Pincetl, T. Gillespie, G. D. Jenerette, and H. R. McCarthy. 2015. Understanding preferences for tree attributes: the relative effects of socio-economic and local environmental factors. *Urban Ecosystems* 18:73–86.
- Buyantuyev, A., and J. G. Wu. 2012. Urbanization diversifies land surface phenology in arid environments: interactions among vegetation, climatic variation, and land use pattern in the Phoenix metropolitan region, USA. *Landscape and Urban Planning* 105:149–159.
- Childers, D. L., M. L. Cadenasso, J. M. Grove, V. Marshall, B. McGrath, and S. T. A. Pickett. 2015. An ecology for cities: a transformational nexus of design and ecology to advance climate change resilience and urban sustainability. *Sustainability* 7:3774–3791.
- Cook, E. M., S. J. Hall, and K. L. Larson. 2012. Residential landscapes as social-ecological systems: a synthesis of multi-scalar interactions between people and their home environment. *Urban Ecosystems* 15:19–52.
- Essl, F., S. Dullinger, W. Rabitsch, P. E. Hulme, P. Pysek, J. R. U. Wilson, and D. M. Richardson. 2015. Historical legacies accumulate to shape future biodiversity in an era of rapid global change. *Diversity and Distributions* 21:534–547.
- Faeth, S. H., C. Bang, and S. Saari. 2011. Urban biodiversity: patterns and mechanisms. *Urban biodiversity: patterns and mechanisms. Annals of the New York Academy of Sciences* 1223:69–81.

- Fuller, R. A., K. N. Irvine, P. Devine-Wright, P. H. Warren, and K. J. Gaston. 2007. Psychological benefits of greenspace increase with biodiversity. *Biology Letters* 3:390–394.
- Grace, J. B. 1999. The factors controlling species density in herbaceous plant communities: an assessment. *Perspectives in Plant Ecology, Evolution and Systematics* 2:1–28.
- Gray, A. J. 1986. Do invading species have definable genetic characteristics. *Philosophical Transactions of the Royal Society of London Series B: Biological Sciences* 314:655–674.
- Grimm, N. B., S. T. Pickett, R. L. Hale, and M. L. Cadenasso. 2017. Does the ecological concept of disturbance have utility in urban social–ecological–technological systems? *Ecosystem Health and Sustainability* 3:1–18.
- Grimm, N. B., R. W. Sheibley, C. L. Crenshaw, C. N. Dahm, W. J. Roach, and L. H. Zeglin. 2005. N retention and transformation in urban streams. *Journal of the North American Benthological Society* 24:626–642.
- Grove, J. M., D. H. Locke, and J. P. M. O’Neil-Dunne. 2014. An ecology of prestige in New York City: examining the relationships among population density, socio-economic status, group identity, and residential canopy cover. *Environmental Management* 54:402–419.
- Grove, J. M., A. R. Troy, J. P. M. O’Neil-Dunne, W. R. Burch, M. L. Cadenasso, and S. T. A. Pickett. 2006. Characterization of households and its implications for the vegetation of urban ecosystems. *Ecosystems* 9:578–597.
- Hall, S. J., R. A. Sponseller, N. B. Grimm, D. Huber, J. P. Kaye, C. Clark, and S. L. Collins. 2011. Ecosystem response to nutrient enrichment across an urban airshed in the Sonoran Desert. *Ecological Applications* 21:640–660.
- Hope, D., C. Gries, W. X. Zhu, W. F. Fagan, C. L. Redman, N. B. Grimm, A. L. Nelson, C. Martin, and A. Kinzig. 2003. Socioeconomics drive urban plant diversity. *Proceedings of the National Academy of Sciences* 100:8788–8792.
- Jenerette, G. D., et al. 2016. Climate tolerances and trait choices shape continental patterns of urban tree biodiversity. *Global Ecology and Biogeography* 25:1367–1376.
- Johnson, A. L., E. C. Tauzer, and C. M. Swan. 2015. Human legacies differentially organize functional and phylogenetic diversity of urban herbaceous plant communities at multiple spatial scales. *Applied Vegetation Science* 18:513–527.
- Kinzig, A. P., P. Warren, C. Martin, D. Hope, and M. Katti. 2005. The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. *Ecology and Society* 10:23.
- Knapp, S., L. Dinsmore, C. Fissore, S. E. Hobbie, I. Jakobsdottir, J. Kattge, J. Y. King, S. Klotz, J. P. McFadden, and J. Cavender-Bares. 2012. Phylogenetic and functional characteristics of household yard floras and their changes along an urbanization gradient. *Ecology* 93:S83–S98.
- La Sorte, F. A., et al. 2014. Beta diversity of urban floras among European and non-European cities. *Global Ecology and Biogeography* 23:769–779.
- Larsen, L., and S. L. Harlan. 2006. Desert dreamscapes: residential landscape preference and behavior. *Landscape and Urban Planning* 78:85–100.
- Larson, K. L., D. Casagrande, S. L. Harlan, and S. T. Yabiku. 2009. Residents’ yard choices and rationales in a desert city: social priorities, ecological impacts, and decision tradeoffs. *Environmental Management* 44:921–937.
- Larson, K. L., J. Hoffman, and J. Ripplinger. 2017. Legacy effects and landscape choices in a desert city. *Landscape and Urban Planning* 165:22–29.
- Luck, G. W., L. T. Smallbone, and R. O’Brien. 2009. Socio-economics and vegetation change in urban ecosystems: patterns in space and time. *Ecosystems* 12:604–620.
- Maas, J., R. A. Verheij, P. P. Groenewegen, S. de Vries, and P. Spreeuwenberg. 2006. Green space, urbanity, and health: How strong is the relation? *Journal of Epidemiology and Community Health* 60:587–592.
- Martin, C. A., P. S. Warren, and A. P. Kinzig. 2004. Neighborhood socioeconomic status is a useful predictor of perennial landscape vegetation in residential neighborhoods and embedded small parks of Phoenix, AZ. *Landscape and Urban Planning* 69:355–368.
- Mayer, F. S., and C. M. Frantz. 2004. The connectedness to nature scale: a measure of individuals’ feeling in community with nature. *Journal of Environmental Psychology* 24:503–515.
- Mulhouse, J. M., L. M. Hallett, and S. L. Collins. 2017. The influence of seasonal precipitation and grass competition on 20 years of forb dynamics in northern Chihuahuan Desert grassland. *Journal of Vegetation Science* 28:250–259.
- Oksanen, J., et al. 2016. *vegan: community Ecology Package*. R package version 2.4-1. <https://CRAN.R-project.org/package=vegan>
- Palmer, M., et al. 2004. Ecology for a crowded planet. *Science* 304:1251–1252.
- Pickett, S. T. A., and M. L. Cadenasso. 2009. Altered resources, disturbance, and heterogeneity: a framework for comparing urban and non-urban soils. *Urban Ecosystems* 12:23–44.

- Pickett, S. T. A., J. Kolasa, J. J. Armesto, and S. L. Collins. 1989. The ecological concept of disturbance and its expression at various hierarchical levels. *Oikos* 54:129–136.
- Pickett, S. T. A., and P. S. White. 1985. *The ecology of natural disturbance and patch dynamics*. Academic Press, New York, New York, USA.
- Pinheiro, J., D. Bates, S. DebRoy, D. Sarkar, and R Core Team. 2017. nlme: linear and nonlinear mixed effects models. R package version 3.1-128. <https://CRAN.R-project.org/package=nlme>
- R Development Core Team. 2015. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
- Radeloff, V. C., R. B. Hammer, S. I. Stewart, J. S. Fried, S. S. Holcomb, and J. F. McKeefry. 2005. The wildland-urban interface in the United States. *Ecological Applications* 15:799–805.
- Reichard, S. H., and P. White. 2001. Horticulture as a pathway of invasive plant introductions in the United States. *BioScience* 51:103–113.
- Rejmanek, M., and D. M. Richardson. 1996. What attributes make some plant species more invasive? *Ecology* 77:1655–1661.
- Ripplinger, J., J. Franklin, and S. L. Collins. 2016. When the economic engine stalls – A multi-scale comparison of vegetation dynamics in pre- and post-recession Phoenix, Arizona, USA. *Landscape and Urban Planning*. <https://doi.org/10.1016/j.landurbplan.2016.05.009>
- Roach, W. J., J. B. Heffernan, N. B. Grimm, J. R. Arrow-smith, C. Eisinger, and T. Rychener. 2008. Unintended consequences of urbanization for aquatic ecosystems: a case study from the Arizona desert. *BioScience* 58:715–727.
- Roberts, D. 2016. labdsv: ordination and multivariate analysis for ecology. R package version 1.8-0. <https://CRAN.R-project.org/package=labdsv>
- Seto, K. C., and R. K. Kaufmann. 2003. Modeling the drivers of urban land use change in the Pearl River Delta, China: integrating remote sensing with socioeconomic data. *Land Economics* 79:106–121.
- Shochat, E., P. S. Warren, S. H. Faeth, N. E. McIntyre, and D. Hope. 2006. From patterns to emerging processes in mechanistic urban ecology. *Trends in Ecology and Evolution* 21:186–191.
- Ter Braak, C. J. F. 1986. Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. *Ecology* 67:1167–1179.
- Tzoulas, K., K. Korpela, S. Venn, V. Yli-Pelkonen, A. Kazmierczak, J. Niemela, and P. James. 2007. Promoting ecosystem and human health in urban areas using Green Infrastructure: a literature review. *Landscape and Urban Planning* 81:167–178.
- Uren, H. V., P. L. Dzidic, and B. J. Bishop. 2015. Exploring social and cultural norms to promote ecologically sensitive residential garden design. *Landscape and Urban Planning* 137:76–84.
- Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan. 2005. The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society* 24:706–723.
- Ward, J. H. 1963. Hierarchical grouping to optimize an objective function. *Journal of the American Statistical Association* 58:236–244.
- Yabiku, S. T., D. G. Casagrande, and E. Farley-Metzger. 2008. Preferences for landscape choice in a Southwestern desert city. *Environment and Behavior* 40:382–400.
- York, A. M., M. E. Smith, B. W. Stanley, B. L. Stark, J. Novic, S. L. Harlan, G. L. Cowgill, and C. G. Boone. 2011. Ethnic and class clustering through the ages: a transdisciplinary approach to urban neighbourhood social patterns. *Urban Studies* 48: 2399–2415.

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