HUMAN-INDUCED VEGETATION DYNAMICS AND AVIAN RESPONSES IN A RAPIDLY EXPANDING DESERT METROPOLIS

by

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ABSTRACT

Burgeoning population growth alongside a vastness of inexpensive land in the American Desert Southwest has generated a culture of unprecedented urban expansion. This dissertation is a study of the ecological effects of urbanization in one of the fastest growing metropolitan areas in the world, Phoenix, combining theories, methods, and data from the ecological and social sciences. Plant species diversity patterns in the desert ecosystem result from abiotic sorting but not in urban sites. Species richness per plot was higher for desert sites; however, the estimated species pool in the urban ecosystem is higher than in the desert resulting from the enhanced importation of introduced species through the nursery trade. A remote sensing classification scheme to extract woody vegetation cover from high-resolution aerial photography was developed and tested generating an urban forest map of the region. This urban forest map was analyzed with sociological census data to study the interplay among economics and patterns of residential development and their effects on canopy cover. Results showed higher canopy cover for older neighborhoods, indicating a detectable temporal lag of urban forest development. Also, more affluent neighborhoods tended to have higher canopy cover, independent of age, even though they are more densely constructed. Lastly, urbanization in a desert ecosystem affects avian communities at two distinct scales. At the regional level, land use configuration affected the distribution of native and exotic species. While exotic species were found to be isolated to the city; native species actively utilized the entire region, even occurring at higher densities in the city than in some areas of the desert. This approach was also used to compare distributional patterns of foraging guilds of birds. At a local scale, the abundance of native species, exotic species and how the foraging guilds of birds responded to vegetation cover measured at varying spatial scales were examined. Bird guilds responded to vegetation at different scales, depending on their association with vegetation.

For Grandma

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CHAPTER 1

INTRODUCTION

Due to our elevated impact on the world's ecosystems, it has become apparent that ecological theory must incorporate humans (Vitousek et al. 1997). Indeed, modern urban development provides an excellent laboratory to examine the interplay among socio-ecological relationships (Grimm et al. 2000). In this system, humans are the predominant controlling factor affecting the majority of ecological processes. Integrating humans into ecological models is challenged by our unique combination of intense social behavior and enormous potential to alter ecological function. This effect is regulated by the interplay of institutionalization, individual action, and traditional ecological dynamics. In order to comprehend this complexity, it is crucial than an understanding of theories and methodologies from many disciplines be integrated and applied (Cronon 1991, Grimm et al. 2000, Machlis et al. 1997, Pickett et al. 1997, Naveh 2000). Historically, ecologists have been slow to accept the urban system as a component worthy of intense ecological investigation (McDonnell and Pickett 1990). This isolated focus has created a suite of ecological theories in which the human component cannot be neatly incorporated (Collins et al. 2000, Padoch 1993, Turner and Meyer 1993). Thus it becomes necessary to link the social and ecological sciences for a more complete understanding of urban ecological dynamics (Pickett and McDonnell 1993, Erlich 1997).

Urban areas occupy 3.5% of the US, contain 75% of the human population, and harbor 3.8 million trees, which cover 27.1% of urban land (Dwyer et al. 2000). Urban vegetation provides numerous ecological and social services such as: enhanced air/water quality (Rowntree & Nowak 1991, Nowak et al. 1998), energy conservation (Heisler

1986, Heisler et al. 1995), climate mediation (McPheson et al. 1994), recreational opportunities, and community well being (Schroeder and Cannon 1983). Urban vegetation affects the human psyche (Ulrich 1981, Ulrich 1984), in which people have expressed “deep emotional ties” (Dwyer et al. 1991). Thus urban vegetation structure, defined here as the spatial configuration and species composition of all plants, in many cities is directly controlled by humans. Vegetation structure defines the biotic matrix in urban ecosystems and directly affects other ecological processes (Clark 2000, Goldstein et al. 1986, Shochat et al. 2004a).

In this dissertation, I analyzed how humans modify vegetation within the Phoenix metropolitan area (Arizona, Untied States of America), and how these alterations affected avian abundance and distributional patterns across the region. The effects of urbanization on vegetation within the Phoenix metropolitan area are analyzed in two ways: biodiversity and spatial configuration. Chapter 2 is an analysis of how the rapidly urbanizing Phoenix metropolis has affected plant species diversity and community composition by comparing the differences at a regional level among urban, agriculture and Sonoran Desert sites, as well as looking at the intra-urban variability of biodiversity. Chapter 3 reveals a classification scheme developed to produce a high-resolution, spatially explicit vegetation map of the Phoenix metropolitan area and surrounding agricultural fields and Sonoran Desert utilizing state of the art remote sensing techniques developed for high-resolution aerial photography. Chapter 4 utilizes this urban vegetation map as a dependent variable with which to analyze the interplay among socioeconomics, patterns of urban development, and their effect on vegetation abundance within residential neighborhoods of the Phoenix metropolitan area. Finally, chapter 5 is an analysis of avian habitat selection at two distinct scales, which explores large scale patterns of distribution of functional groups alongside a scalar analysis of avian dependence (or lack thereof) on vegetation at variety of local scales.

CHAPTER 2

A TRANSPORTED OASIS: PATTERNS AND CONTROLS OF PLANT DIVERSITY IN AN URBANIZING DESERT METROPOLIS

Edgar Anderson proclaimed that “man carries whole floras about the globe, that he now lives surrounded by transported landscapes” (1952). With this acknowledgement of our global impact, it has become apparent that ecological theory must incorporate human action (McDonnell and Pickett 1993, Vitousek et al. 1997), and nowhere is the effect of humans greater than in and around the city (Pickett et al. 1997, Grimm et al. 2000). In this chapter, a set of predictions are presented and tested regarding the effects of urbanization on plant diversity in the Phoenix metropolitan area harbored within the Sonoran Desert of central Arizona. Due to its arid nature, plant community assembly in the Sonoran Desert has long been attributed to effects of soil (Shreve 1951). Variability in productivity in deserts is generally attributed to spatial differences in soil moisture availability (Went 1949, Beatley 1969) and limiting resources, especially nitrogen. Soil moisture availability is both a function of the amount of precipitation and the water holding potential of the soil. In the Sonoran Desert, precipitation corresponds to plant productivity and increases with elevation (Sheppard et al. 2002). Furthermore, productivity in desert systems tends to be positively correlated to species richness (Boeken and Shachak 1994). While the determinants of plant communities in the Sonoran Desert have been well studied, the effects of humans are less well understood.

Anthropogenic drivers of ecosystem change via urbanization operate as both top-down and bottom-up controls (Figure 1). In an urbanizing system, top-down regulation of vegetation dynamics is initiated by land-use change. This process is typically preceded by zoning by a governmental agency, which determines how and where development will occur. Zoning affects how an area will change by determining its ultimate land-use type (i.e. a neighborhood, business, or street). The next step in urban development involves architects and urban planners, who decide where the built components of the city will be located. Their decisions determine where the spaces between the buildings and the roads will be, which correspond to the potential sites of plant growth. Thus, spatial distribution of impervious surface as a result of the urbanization process should dictate the density of plants that can exist.

Following urban design, lots to be built are cleared for development, usually by wholesale removal of vegetation (Rebele 1994). Once the urban infrastructure (e.g., roads & buildings) becomes established, plant communities are reconstructed by landscape architects and assembled by landscaping installation companies. These companies acquire vegetation stock from the nursery trade, which is driven by markets that respond to horticultural aesthetics. Such nursery companies diversify their investments by filling their greenhouses with many varieties of showy plants, most of which are exotic species. Thus, this diversification should increase the species pool of the urban ecosystem above that of its desert counterpart, populated by an abundance of exotic species. Another form of top-down regulation of plant diversity occurs from neighborhood homeowner's associations (HOAs), whose governing rules can limit the species pool (Martin et al. 2003). Zoning, construction, design, markets, and HOA decisions comprise the primary top-down anthropogenic forces that determine patterns in urban plant diversity and abundance.

Bottom-up anthropogenic forces also affect such patterns at a household scale. In an earlier study on plant diversity in Phoenix, Hope and others (2003) reported a socalled “luxury effect”, in which numbers of perennial genera in wealthy neighborhoods were greater than in poor neighborhoods, which they attributed to additional disposable income of the former. We build upon this earlier work using a richer data set of plant diversity, and we propose a mechanism to explain patterns of plant diversity and community composition within the Phoenix metropolitan area that is based upon a combination of socioeconomics and landscaping choices.

Phoenix’s urban ecosystem is unique relative to many other American cities in that three disparate landscaping styles have developed (Martin 2001, Martin et al. 2003). Prior to the advent of the air conditioner, the most common home landscape was *mesic*. This landscaping type was originally supported by flood irrigation and featured shade trees and flowering shrubs planted in lawns for the purpose of reducing latent heat around living spaces (Folkner 1958). These mesic landscapes emulated the landscape aesthetic, but not necessarily the species pool, common to the eastern USA. Once air conditioners became common, Phoenicians were afforded the climatic luxury of *xeric* landscaping, characterized by low water-use plants, such as desert-adapted trees, shrubs and succulents, planted in inorganic mulch such as crushed rock (Martin 2001). Martin and others (2003) recognize a third common landscaping type, *oasis*, which merges elements of the xeric and mesic palette and is typified by ground cover including both inorganic mulch and lawn within the same landscape. For pictographic examples, refer to Figure 2. It is anticipated that this differentiation in landscaping aesthetic, which is largely a result of choices made on the household scale, will produce distinct differences in both patterns of plant diversity and community composition within the urban ecosystem.

The general hypothesis is that plant diversity and community composition in an urbanizing area are controlled at different scales by top-down and bottom-up factors. Regionally, land-use change during urbanization is a top-down anthropogenic effect, directly altering plant diversity. Within the desert, diversity is typically explained by bottom-up, geophysical factors. Within urban areas, both top-down and bottom-up factors control diversity and community composition. However, urban areas differ from the desert in that controls derive primarily from human action such as access to economic resources and landscaping choices, rather than geophysical factors.

Within this chapter the following predictions are tested:

P_1: The urban ecosystem is a "transported landscape" characterized by the removal of native species and promotion of introduced species. In the urban ecosystem, the encouragement of introduced species through the nursery trade is so intense that the species pool is greater in urban than in desert locations.

P_2: Biodiversity within desert sites is significantly correlated with abiotic factors such as soil texture, geomorphic, and geochemical variables; whereas biodiversity in urban sites is not related to these variables.

P_3: Urbanization results in expansion of built structure, including buildings, infrastructure, and impervious surfaces with a concomitant reduction in growing space for plants. Thus, at local scales urban areas have reduced plant density, and correspondingly reduced plant richness, compared to desert areas.

P_4: Within the urban ecosystem, variations in plant diversity and community composition are a function of bottom-up forces. Plant species richness will be positively correlated with income and plant community composition will differ among unique landscaping regimes (e.g. xeric vs. mesic).

Methods

Study area and Sampling Procedure

During the latter half of the twentieth century, the Phoenix metropolitan area (hereafter "Phoenix") of central Arizona (Figure 3a) has seen exponential growth driven by its favorable climate, cheap housing and water, economic opportunity, and especially the commercial success of the air conditioner (Luckingham 1989). Now harboring nearly five million people, Phoenix is the fifth largest and the fastest growing metropolitan area in the nation (U.S. Department of Commerce, Bureau of the Census, 2001). As part of the Central Arizona-Phoenix Long-Term Ecological Research (CAP LTER) program, ecological surveys were conducted at 204 sites covering 6400 km^2 encompassing the entire Phoenix metropolitan area, including agricultural lands and undeveloped Sonoran Desert (Figure 3a). Permanent site locations were identified by a dual-density, randomized, tessellation-stratified, sampling regime (originally described by Hope et al. 2003). A field inventory was conducted between February and May, 2005 following the same protocol as that conducted in 2000 (Hope et al. 2003, 2006). The sampling unit at each site was a 30m x 30m (900 m^2) plot, in which plants of all taxa were identified, collected, and archived at Arizona State University's herbarium. All perennial plants were counted within the plot in order to obtain density (perennials/900 m^2). Density of annual plants was estimated as the linear proportion along two 30-m perpendicular lines that dissected the plot exactly into quarters.

Following taxonomic identification, taxa were grouped into a variety of morphological or functional groupings based on Kearney and Peebles (1951) and USDA classifications (USDA 2005): (1) native (to central Arizona) vs. introduced species, and (2) annual vs. perennial species. Species richness (S) was calculated as taxa/plot, and evenness (E) of perennial taxa was calculated according to Hayek and Buzas (1997):

$$E = e^{H'} S^{-1} \qquad \text{(Eq. 1)}$$

where *H'* is Shannon-Weiner's diversity index.

The main surface-cover types (asphalt, concrete, bare soil, turf) on the 900-m^2 plot were mapped, and these maps were used to determine proportion impervious surface within the plot. Land use at each of the sites was classified into four regional categories: (1) *urban* (n= 95 sites), (2) *desert* (n=74), (3) *agriculture* (n=25), and (4) a *mixed* class (n=12). Within urban sites, further subdivisions were (1) *residential* (n=54), (2) *commercial-institutional* (n=16), (3) *transportation* (n=13), (4) *vacant-under construction* (n=8), and (5) *mixed* (n=4). Finally, within urban sites, landscaping typology was also assessed on-site and categorized as (1) mesic (n=30), (2) xeric (n=26), (3) oasis (n=18), and (4) unclassified (n=21). *Xeric* plots always lacked turf lawns and were usually covered by inorganic mulch (i.e. gravel); *mesic* plots had a large expanse of lawn and always lacked gravel mulch; and *oasis* was some combination of these two. *Unclassified* sites did not clearly fall into the landscaping choice categories. Elevation, slope, and aspect were extracted from an overlain USGS Digital Elevation Model (30 m resolution) for each of the sample locations. Four soil cores (30 cm deep, data from upper 10 cm were used for this analysis) per site were taken, composited, and composites were analyzed for N, P, C and texture (Hope et al. 2005; Zhu et al. 2006). Income per capita was obtained from the U.S. Census (2001) for the block group surrounding each *residential* survey point.

Analyses

Analysis of variance was conducted (Figure 3) for two complementary metrics of diversity (richness and evenness) and density (perennial and annual species) with regional land-use categories (urban, desert, and agriculture) as factors. Differences in plant richness were also analyzed, with urban land-use type (commercial, vacant, transportation, and residential), and landscaping categories (xeric, mesic, and oasis) as factors. Sites considered *mixed* between land uses were omitted for clarity. To determine whether there was an effect of abiotic factors on plant diversity (richness and evenness), factor analysis with varimax rotation was conducted with the abiotic variables to achieve independent eigenvectors, which were subsequently run in stepwise regression models with two subsets of the data (urban and desert). In cases that satisfied the assumption of normality and constant variance for the ANOVA model, overall significance was tested by the *F* test for equality of factor levels. In cases where the assumptions could be satisfied by transformations of the variables, as suggested by the Box-Cox procedure (Box & Cox 1964), the *F* test was conducted on the transformed variables. For all cases where the *F* test was appropriate, multiple pairwise comparisons were conducted using the Tukey-Kramer method (Kramer 1956) with $\alpha = 0.05$. Where a transformation could not satisfy assumptions of normality and/or constant variance, the Kruskal-Wallis test (Kruskal & Wallis 1951) was utilized to test for equality of the factor levels, and multiple comparisons were conducted using Dunn's method (Dunn 1964). Sample-based rarefaction was conducted to interpolate species accumulation curves for each of the regional land-use classes (Figure 3f), estimated as S_{exp} (the expected species richness for a given sample; Colwell et al. 2004; Colwell 2005). Ordination via non-metric, multidimensional scaling (NMS) was employed to suggest patterns in community structure (Clarke 1993; McCune & Grace 2002). We adopted this approach to address whether plant communities formed assemblages based on the abovementioned land-use and landscaping categories (Figure 4).

Results and Discussion

Abiotic controls of desert plant diversity

We predicted (P_1) that variability in biodiversity within the desert could be explained by abiotic factors (i.e., soil texture, geomorphic and biogeochemical variables); whereas biodiversity in urban sites could not. In order to analyze the effect of these variables on plant diversity, forward stepwise regression, with α_{in}=0.05, was conducted separately on diversity measurements from two subsets of the data, desert only and urban only, on the eigenvectors from the factor analysis from the following abiotic variables: elevation, slope, aspect, N, C_{inorg}, C_{org}, P, and the proportions of sand, silt and clay. As predicted, none of these variables explained a significant portion of the variance in taxa richness or evenness among urban sites. However, in desert sites, 20% of the variability in plant richness was significantly explained by a factor largely associated with elevation, and 6% of the variability in evenness was significantly explained by that same factor. Within desert sites, the positive influence of elevation on plant richness may be attributed to increases in rockiness with increasing elevation. Plant-available water is enhanced via condensation and reduced evaporation between rocks and in rock fissures. This increases productivity and density, factors which also tend to be positively correlated with species richness in arid regions (Rosenzweig 2002). In addition, increasing substrate heterogeneity at higher elevations may promote co-existence by creating independent microsites, thereby minimizing competitive exclusion by dominants (Levins 1969, Tilman 1994). This supports the prediction (P_1) that abiotic variables traditionally used in the Sonoran Desert to explain plant diversity are still valid in the desert adjacent to a large metropolis. However, these variables are not significant determinants of plant diversity within the urban system.

Urbanization as a transported landscape: top-down effects

Urbanization has a dramatic effect on plant abundance and biodiversity patterns, generated by a combination of top-down and bottom-up anthropogenic effects. At the plot scale, urbanization has decreased species richness while enhancing plant community evenness (Figure 3d-e). These results suggest urbanization homogenizes the plant community at local scales, creating a community that consists of fewer species and with higher evenness, relative to desert plots. Analysis of species-accumulation curves provides a different insight at the regional scale (Figure 3f), revealing a plant species pool that is more speciose within the urban area than in the desert, as predicted (P_2). Enhancement of the urban species pool can be largely attributed to introduction of plant species, purposefully and incidentally, into agricultural and urban areas. The higher proportion of exotic species and lower proportion of native species within the urban ecosystem (Figure 5a) is consistent with this interpretation. Thus, plot-scale species richness is highest for desert sites; but the cumulative species richness, or species pool (also known as γ-diversity), is highest for urban sites due to promotion of introduced species by urban residents and landscapers.

Land-use change not only affects patterns of plant diversity, but also patterns of plant community composition. Ordination of species' occurrence coded by regional land-use type yields three unique plant communities (Figure 4a). Such groupings suggest that at a regional scale, there is a dominant, top-down effect of land-use change, which clearly alters not only plant diversity, but also plant community composition with very clear consequences. On further examination of the ordination, those urban sites that overlap with desert sites are mostly sites that occur at the urban fringe, suggesting an urban to rural ecotone of plant community composition. More recent housing developments on the urban fringe often are constructed without complete removal of vegetation, in contrast with the older, interior urban sites, resulting in the overlap in plant community composition between fringe and desert sites. In contrast to the regional land-use analysis, ordination on different land uses *within* the urban system (e.g., residential, commercial) reveals no clear plant assemblages (Figure 4b). Thus an urban plant community exists, but assemblages are not unique to different urban land types. Thus, land-use change is a significant top-down factor on plant community composition only at the coarsest scale. Regarding the legacy of land use transformation on plant communities, it does not appear to matter what type of urban land use it will become, just that it will become urban.

The most dramatic effect of urbanization on plants is reduction in plant density (Figure 3g-h). One factor that may be driving this pattern is lower availability of growing space in the city. To evaluate this, a regression was conducted with the proportion impervious surface coverage, which should be inversely related to available growing space for plants, with plant density. Within the urban sites, no significant relationship was found between plant richness (p =0.46) or the density of perennial plants (p =0.43) and impervious surface coverage. We did find a significant, though weak, correlation between annual plant density and impervious coverage (p=0.005, r^2=0.10). Thus, although urbanization results in a decrease in diversity and density of plants, this decrease cannot be explained by lack of growing space, contrary to our prediction (P_3) except in the case of annual plants. Water is a strong controlling factor for desert plant productivity, but given the abundance of irrigation water in the urban environment (Martin 2001), we would expect plant density to increase in from an ‘oasis effect.’ However, we found plant density to be lower in the city than in the desert despite the added water, suggesting that other human factors play a primary role in determining plant density. Perhaps, humans are limiting density by favoring fewer, larger specimen plants, an idea that is supported by the fact that woody vegetation coverage is higher in the city than in the desert (Chapter 3).

Household-level effects on the urban plant community

While there is evidence that the process of land transformation from rural to urban produces distinct alterations in patterns of plant diversity and community composition, changes within the urban ecosystem are likely a result of individual action at the household level. In order to analyze household-level effects, correlations between plant species richness and average household income were examined for the surrounding neighborhood, following Hope and others (2003, 2006). Differences were then evaluated in plant community composition among different landscape choices (mesic, oasis, xeric).

Based upon their analysis of data from a similar survey conducted in 2000, Hope and others (2003, 2006) posited a "luxury effect," in which wealthier neighborhoods tended to have a greater number of perennial plant genera (partial $r^2=0.17$ for a multiple regression model, $p=0.0003$). This study was enhanced by identifying all taxa, not just perennial taxa, to the species level where possible. Furthermore, samples were omitted that did not occur within the urban ecosystem. While plant richness did significantly vary with income ($p = 0.01$), income explained an even lower proportion of the variance than in the 2000 studies ($r^2=0.09$ compared with 0.17). It is important to note that the relationship between income and species richness was weaker with inclusion of annual species, which were not incorporated in the Hope et al. (2003) analysis. The relationship between income and perennial species in the present (2005) survey was nearly identical to that of the 2000 survey ($p=0.0006$, $r^2=0.17$). Thus, a luxury effect remains evident even with a finer resolution of taxonomic classification, but that luxury effect is stronger for perennial species alone than for all species of plants combined.

While no clear plant assemblages by land-use type emerged from the ordination *within* the urban ecosystem (Figure 4b), it was anticipated that the different landscaping choices (*mesic*, *oasis* and *xeric*) would produce unique plant assemblages. Ordination based on landscaping choices reveals two unique groupings (Figure 4c); with mesic landscapes forming an assemblage of plant taxa distinct from that of xeric and mixed landscapes. Plant assemblages of oasis landscapes were indistinguishable from xeric landscapes, so the existence of lawn had little effect on overall community structure for such landscapes. Furthermore, xeric sites promote native richness and inhibit introduced species to a greater extent that mesic sites (Figure 5). Collectively, bottom-up, household-level factors of family income and landscaping choice appear to be more robust predictors of plant diversity, density, and composition within the urban ecosystem than the specific type of urban land use, supporting the prediction, P_4.

Conclusions

Plant diversity and community composition are driven at a regional scale by the regional process of land-use transformation during urbanization and corresponding importation and promotion of introduced species - largely through the nursery trade. At more local scales, diversity within desert sites is largely a product of biophysical controls by abiotic variables. However, at local scales within the urban ecosystem, variations of diversity and density of plant species are driven by a combination of bottom-up, anthropogenic forces, such as landscaping choice and socioeconomics. Deconstructing this complexity is challenging but necessary. This will require the synthesis of data and methods from both the social and natural sciences, and the evolution of new, blended theories that can account for both ecological processes and human management.

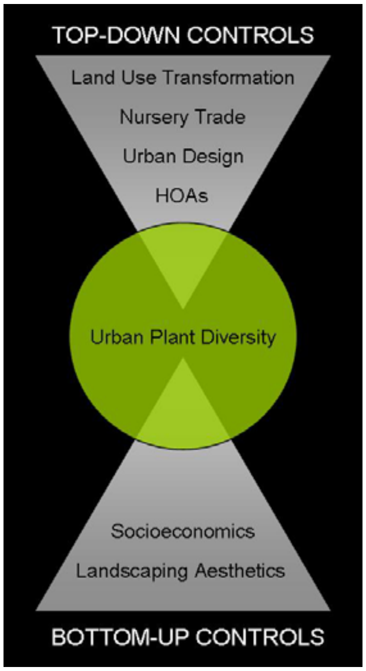

FIG. 1. Conceptual model of the top-down vs. bottom-up effects of urbanization on plant diversity and community composition within the city.

(a) Sonoran Desert (b) Agriculture (c) Transportation

(d) Commercial (e) Vacant/Construction

(f) Residential (mesic) (g) Residential (oasis) (h) Residential (xeric)

FIG. 2. Typical land use types and landscaping typologies used in this analysis.

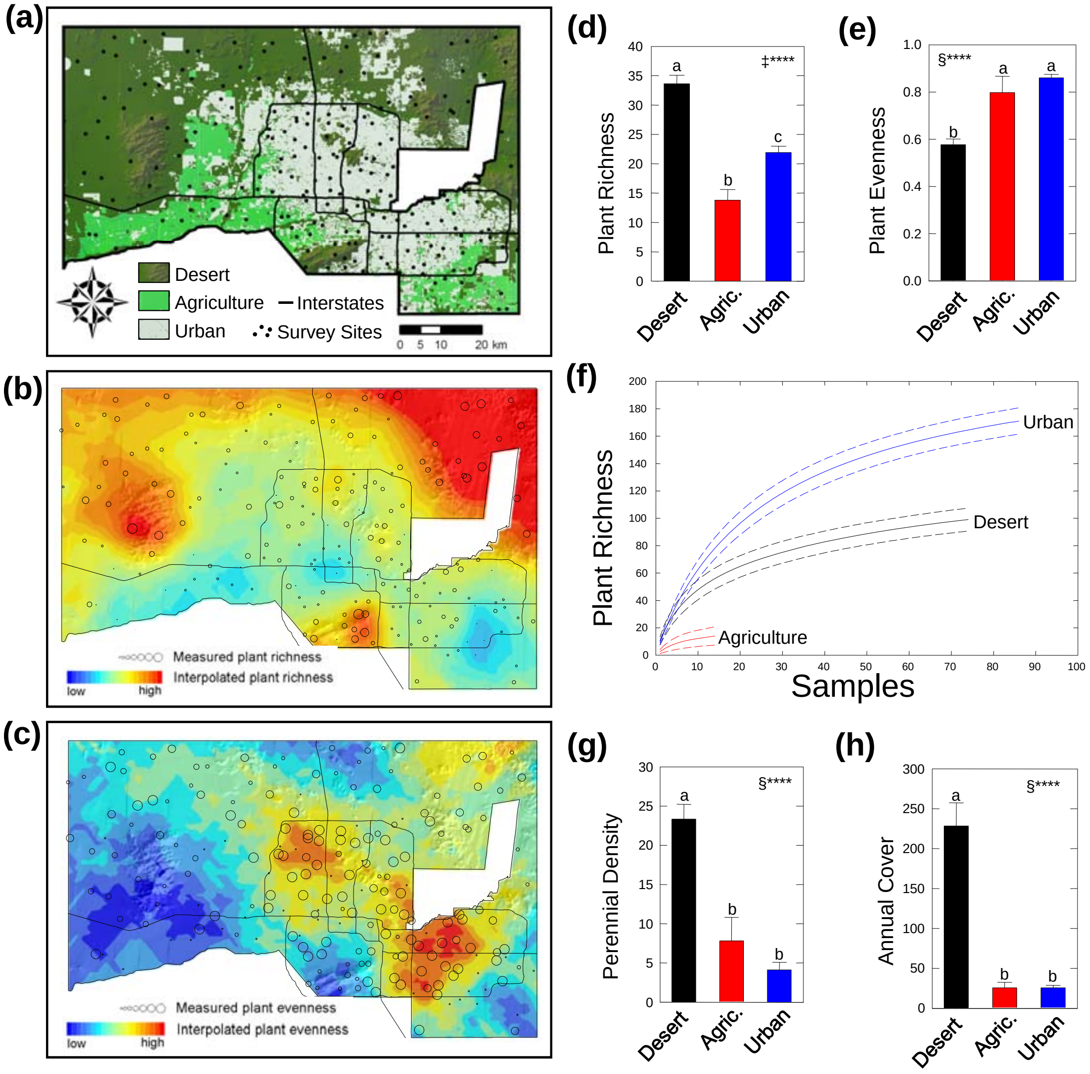

FIG. 3. Regional land use map of CAP LTER and survey site locations (a). Measured and interpolated plant richness (b) and evenness (c). Interpolations were conducted using ordinary kriging. Multiple pairwise comparisons were conducted on plant diversity and density across three regional land use types: desert, agriculture, and urban. Diversity was measured as (d) richness (taxa/900 m^2) and (e) evenness. Species pools were estimated (f) for each of the land uses with EstimateS (Colwell, 2005). Dashed lines represent the 95% confidence intervals. Density was measured as: (g) perennial density (individuals/900m^2)

and (h) annual coverage (m/60m). ANOVA was conducted by the F test[‡] if assumptions of that test were met. Otherwise ANOVA was conducted by the Kruskal-Wallis test[§]. Letters refer to groups that were different according to either Tukey-Kramer's test[‡] or Dunn's test[§], respectively. $\alpha=0.05$ for all tests.

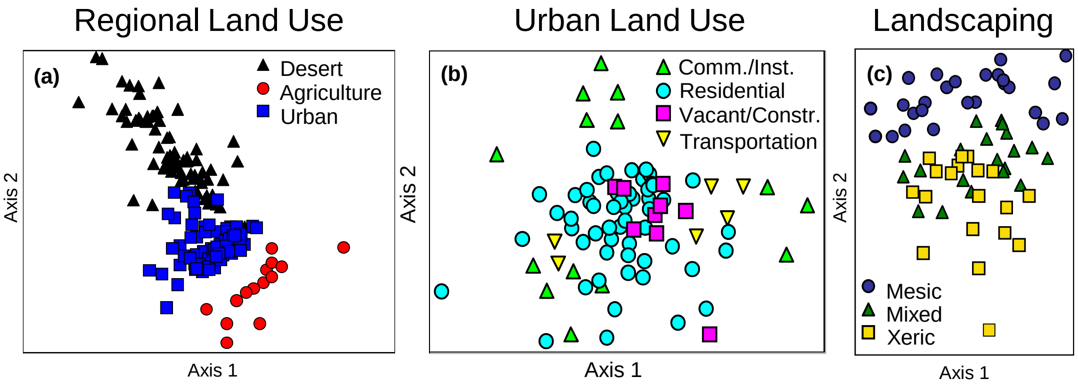

FIG. 4. Plant community ordination via non-metric multidimensional scaling.

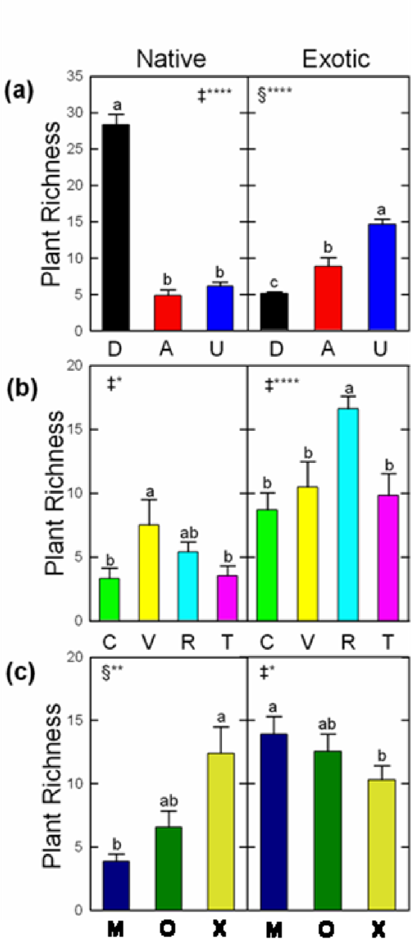

FIG. 5. Multiple pairwise comparisons of native and exotic plant richness across regional (a) and urban (b) land use types and landscaping aesthetics (c) within urban areas. Regional land use types: desert (D), agriculture (A), & urban (U). Urban land use types: commercial/institutional (C), vacant/under construction (V), residential (R), and transportation (T). Landscaping aesthetics: mesic (M), oasis (O), and xeric (X). Statistical tests follow Figure 1.

CHAPTER 3

AN OBJECT-ORIENTED APPROACH TO HIGH RESOLTUION URBAN FOREST MAPPING IN PHOENIX

It has become apparent that human-dominated ecosystems need to be analyzed at broad spatial and temporal scales with sufficient detail in order to assess how humans are modifying their environment (Allen and Hoekstra 1992, Zipperer et al. 1997). Remote sensing can be used to quantitatively collect this information ranging from site-specific locations to the entire landscape. However, urban systems present particular challenges. Scale mismatch between patch and pixel within the urban ecosystem is a special challenge due to the discrete heterogeneity of the system (Iverson et al. 1989, Ridd 1995; Nowak et al. 1998).

Since the inception of Landsat over three decades ago, satellite imagery has been used to evaluate human impacted ecological function across regional and global scales (Forster 1985). Most analyses of urban land cover over this time period have been restricted by coarse resolutions, confining interpretation of these analyses to generalities. For example, an examination of the United States' urban forests has been conducted with AVHRR imagery (1.1 km spatial resolution). This assessment of the nation's urban forests concedes, "data on urban forest ecosystem complexity and variability at the local scale unfortunately are limited" (Dwyer et al. 2000). Local scale phenomena tend to greatly influence patterns of urban land cover, creating a patchwork of fine-scale heterogeneity characteristic of heavily human dominated ecosystems, and it is has become critical that remote sensing data be collected at a finer scale (Carlson 2003).

In order to understand the dynamics of how humans impact vegetation abundance and distribution during the process urbanization, it is important to accurately collect spatially explicit information on vegetation abundance (McPherson et al. 1997). This important land cover type directly affects many phenomena of the urban ecosystem (McBride & Jacobs 1986). Urban vegetation provides numerous ecological, environmental and social services including: enhanced air/water quality (Rowntree & Nowak 1991, Nowak et al. 1998), energy conservation (Heisler 1986, Heisler et al. 1995, Gallo et al. 1993), climate mediation (McPherson et al. 1994), recreational opportunities, and community well-being (Schroeder and Cannon1983). Urban vegetation is unique from rural vegetation. Trees in many exurban ecosystems tend to be in groups; whereas urban trees are typically isolated individuals. Thus, analysis of high spatial resolution imagery is key to understanding the impacts of urban vegetation on its ecosystem (Avery & Berlin 1992).

In this chapter, a methodological technique is presented to allow for regular monitoring of structural vegetation change for the total extent of the Phoenix metropolitan area using high-resolution, true-color ($0.6m^2$) imagery.

Methods

Imagery

True-color (red, 650 nm; green, 510 nm; and blue, 475 nm) digital aerial images (Fig. 6) were collected by a Zeiss RMK aerial photography camera with a 153 mm (6”) focal length lens on April 17-19, 2003 for the entire Central Arizona-Phoenix Long Term

Ecological Project (CAP LTER), which covers 6,400 km^2, which includes the Phoenix metropolitan area, including agricultural fields, and large expanses of surrounding desert. The sensor was flown at an altitude of 6,100 m producing an image with approximately 0.61 m spatial resolution with a geographic accuracy of 9 m. The images were obtained orthorectified as a single tile into North American Datum of 1983 State Plane, Central Arizona.

Object-oriented approach

We employed an object-based, methodological design for classifying urban vegetation through a hybrid of image segmentation and classification based on spectral and contextual values (Blaschke & Hay 2001). To more accurately estimate real world objects, the image is apportioned into basic units for analysis at the object-appropriate scale before classification can occur (Baatz & Shäpe 2000, Blaschke et al. 2000). New technology allows for an object-oriented approach through an algorithm for image segmentation, which subdivides the entire photograph into regions at a user-defined scale, in essence creating patches as the fundamental unit of analysis (Fig. 7, Definiens 2003). This approach takes into account inherent information (i.e. within-pixel spectra values and patch texture) as well as neighborhood characteristics making possible the extraction of real-world objects, proper in shape, as the basic units for analysis. These objects can then be classified based on contextual relationships, including spectral signatures, texture, and shape metrics, such as landscape shape index, and elliptical fit. This approach has also been applied to ASTER and Landsat images of Phoenix for more regional analyses of land use change (Möller 2005).

Pre-classification processing: image division and segmentation

The image was divided into 100 km^2 square tiles for analysis due to software limitations on image size due to the processing power required by the segmentation process. These tiles were put through a process of data-driven image segmentation, which subdivides the image into polygonal units at a particular scale based on spatial and spectral values (Ryherd & Woodcock 1996). We employed the fractal net evolution approach (FNEA), an objected-oriented segmentation algorithm embedded in the software eCognition (Baatz & Shäpe 2000, Definiens 2003). The segmentation algorithm is based on three parameters: scale, color, and shape. Scale and color can be weighted from 0 to 1; and shape is divided into two settings, smoothness and compactness, which can be complementarily weighted from 0 to 1. Segmentation parameters employed in this analysis are summarized in Table 1.

The proper level of segmentation is determined by the size of the object in question (Benz 2004). An image can be partitioned at a variety of segmentation levels (or "scale parameters"), and classification of particular objects can be extracted at a user-defined scale. This parameter is directly dependent on the spatial resolution of the data and the relative extent of the objects of interest. If a researcher was interested in broad scale phenomena, such as urban growth models, a very large scale parameter would be used creating few large objects that could be classified into coarse land use types (i.e. urban vs. rural). A much smaller segmentation level might be necessary for analysis of road networks, and even smaller still for houses. Urban vegetation is characterized at the smallest extent of land cover detectable by this imagery, as the coverage of most vegetation in the urban environment is quite small, well below 30m.

Classification

Once segmentation was completed, the image was then analyzed and classified based on the spectral and topological characteristics of the image objects (for classified example see Fig. 8). The spatial resolution of this imagery allowed for active interpretation of samples to develop the classification methodology, creating a user-defined classification for the classes: *WOODY*, indicating objects determined to be a shrub or tree, and *OTHER*, everything else including roads, buildings, soil and grass.

As the samples were selected a variety of descriptive metrics were compiled for each of the image objects including the following groups: (1) the spectral nature, (2) shape characteristics, and (3) texture of the object. All of the groups of metrics were tested for ability to discriminate *WOODY* from *OTHER*.

Shape characteristics are virtually useless when image segmentation is done at such a fine resolution relative to the spatial resolution of the imagery. This creates many shapes of many differently classified things to have the same shape characteristics (see Fig. 7). One would expect vegetation to have a landscape shape index closer to circular than most objects. However, this is not the case due to the segmentation resolution employed. When segmenting an image with such a low scale parameter (high resolution), the image objects of most all land covers are virtually identical; and thus, shape is not a good metric for discrimination.

Determining the texture of the object was considered troublesome for the same reason as the shape characteristics; the texture is very similar in objects of different classes due to the fine resolution segmentation. During the discrimination analysis, we determined that analyzing only the spectral nature of image objects was most desirable, as computations are relatively simple and discrimination was feasible between the two classes.

A feature-space optimization was performed next in order to determine which of the spectral metrics would be most suited for discrimination between the land cover classes. All possible combinations of spectral metrics were analyzed for each band: brightness, mean of all pixels in image object, standard deviation of pixels, and the ratio of mean and brightness. Histograms of samples from both classes were isolated for each of the metrics selected.

These histograms were transposed on top of each other and were analyzed for adequate discrimination. This process produced five metrics useful in discriminating the two classes. They were, in order:

(1) mean red

(2) brightness

(3) mean green

(4) mean blue

(5) mean green:brightness

The selection for these metrics was also supported by considering the low percentage of histogram overlap between the classes (Table 2.). The largest overlap allowed was brightness, defined as “the channel mean value of all image layers” (Definiens 2003). This metric was utilized because shadows appeared very distinct in brightness than other objects in the highest region of the histogram. The first dichotomous decision was “shadow” and “non-shadow” based on the brightness values, 0-0.12 for shadows and >0.12 for everything else. It became apparent that “shadow” also included many water bodies, so this class became “shadow/water”. Everything else in “non-shadow” was then analyzed for distinction between *WOODY* and *OTHER*. Mean red and mean blue were appropriate for discrimination among vegetation and most other objects, mainly pervious surfaces, roads and buildings. Parameters for *WOODY* were DNs of 5-99 for mean red and 5-63 for mean blue. The ratio *mean green:brightness*, with parameters 0.36-0.53 enhanced discrimination power of grass and woody vegetation. The addition of mean green to the classification scheme did not improve discrimination power and was thus not incorporated.

Post-classification processing

The need for high-resolution segmentation generated fine-grained polygons for analysis. However, many instances were encountered where more than one polygon were created for a single object of interest. Adjacent polygons that were classified in the same category were fused by classification-based segmentation within the individual tiles. Neighborhood characteristics were then used to identify object classified as “shadow/water” that shared >40% of its border with another object classified as *WOODY*. Then those selected objects would be reclassified as *WOODY*. This was done because many tall trees were creating a shadow on the other half of tree and improperly classifying it. Those classified tiles were then fused back together in ERDAS Imagine (2004), in essence creating a spatially explicit measurement of woody vegetation of the Phoenix metropolitan area (Fig. 9).

Groundtruthing

To determine the accuracy of the classification, an extensive ground-truthing campaign was conducted (Fig. 10). Two hundred and three sampling locations were randomly selected from the Phoenix metropolitan area defined as *urban* by the Maricopa Association of Governments (2000). The sites were diverse in land use type and included: 123 residential, 26 commercial, 16 vacant land (brownlots), 10 institutional (schools, hospitals, etc.), 10 parks, 7 transportation, 7 golf courses, 4 sites under construction, and 2 industrial.

At each sampling location, the central point was located using a Trimble Pro XRS® GPS unit. The sampling unit was a 30m x 30m plot in which all dominant land cover types were identified and mapped using the GPS with sub-meter precision. This sampling extent was chosen to mirror CAP LTER's Survey 200, an extensive analysis of biotic communities and abiotic parameters across the Phoenix metropolitan area that is repeated every five years (Chapter 2, Hope et al. 2003). Any observations recorded above 1m spatial accuracy were omitted to enhance the precision of the accuracy analysis.

At these sites, 1335 observations were made of which 924 were perennial vegetation and 411 were other land cover types. The following land cover classifications were recorded, (1) *perennial vegetation*, (2) *grass*, (3) *bare soil*, (4) *gravel*, typically used in desert landscaping, (5) *concrete* (light in color), (6) *asphalt* (dark in color), and (7) *buildings*. Observations made of grass were also sub-classified as *dormant* or *overseeded*. With Phoenix's very mild winter, grass can be overseeded during the winter which provides a green lawn through late spring, which may increase errors when attempting to classify woody vegetation. Lawns not overseeded have only yellow, dormant grass exposed. All perennial vegetation, including groundcovers, shrubs and trees, were georeferenced at the center of the canopy. Each plant was identified to genus and the largest diameter of the plant was recorded to the nearest meter.

Results

Error assessment

A confusion matrix (Table 3) was generated by comparing the reference collection with the classified imagery (Congalton 1991, Congalton & Green 1999). The classification system was binary; so the simplest way to analyze the accuracy was to aggregate all non-woody referenced land covers into *OTHER*.

We subdivided that classification based on plant size (diameter) and plant type (genus). Omission errors were abundant in smaller vegetation (Fig. 11). Producer's accuracy was consistently above 0.85 when vegetation was greater than three meters in diameter. This error at the lower size classes may be a result of the segmentation process, due to the aggregation of pixels to formulate image objects as the basic units of classification. The mean pixel number per image object is 8.6 with a variance of 7.3. Thus, there is large variation that is due to the level of heterogeneity of neighboring pixels. It is possible that plants near the size of the resolution of the pixel may be properly segmented or they may be aggregated into an image object of another land cover type (i.e. grass) and then misclassified.

Omission error of the class, *OTHER*, was analyzed using the classification of land cover type of the reference data (Table 5). The only reference points that were improperly labeled as *WOODY* during the classification process were lawns that have been overseeded & asphalt. No omission error occurred in the other referenced classifications. Overseeded lawns are a likely candidate for misclassification, as they consist of dense, healthy vegetation that has a similar reflectance of large, leafy trees. Asphalt has high spectral absorbance similar to vegetation causing cross-classification issues when only looking at the spectral characteristics.

An analysis of commission errors by user's accuracy indicates that if an image object is classified as *WOODY*, there is a 96% probability that it is a woody perennial (Table 3). However, the user's accuracy for *OTHER* is far lower at 0.63. This is due to the misclassification of objects classified as *OTHER* and referenced as *WOODY*.

Discussion

In order to empirically analyze many urban ecological processes it is inherent to rectify the biological matrix of the city by collecting information on plant abundance and location with increasing accuracy and precision (Nowak 1992). Nowak and others present four ways of estimating urban tree cover with appropriate resolution (1996). However, these approaches are either very labor-intensive and/or only sample small portions of the urban ecosystem. The discrete nature of urban landscapes requires the analysis of broad extents at fine resolution. To maximize the benefit of long term ecological monitoring, it is important that methodologies not be labor intensive nor cost prohibitive.

In this chapter, a classification scheme was developed and tested in order to classify aerial photography to highlight elements of the urban forest. The classification was parameterized to maximize the user's accuracy of *WOODY* in order to use the remotely sensed image to map woody vegetation with a high confidence of proper classification. The output of this classification scheme will be used as a spatially explicit measurement of urban forest structure in the following chapters. The goal of this procedure was to produce a reproducible classification procedure that adequately highlights elements of the urban forest.

Examination of omission errors within vegetation type led to some interesting insights (Table 4). Genera that have high producer's accuracy are characteristically larger, exotic trees with dense, coarse foliage, typical in many landscaping practices. Genera with lower producer's accuracy are typically dominated by desert adapted species, many of which are native. These plants are likely not well-classfied for three reasons. Many are small in diameter (i.e. *Washingtonia*, *Cylindropuntia*). Others have diffuse canopies causing most of the pixel to be saturated with the underlying substrate (i.e. *Fouquiera, Larrea, Caesalpina*). Some have an arid adaptation of highly reflective leaves that appear gray in color (i.e. *Agave, Leucophyllum*, *Cassia*). All of these characteristics confuse the classification of the image objects and lead to error.

Grass can be very spectrally variable especially in spring, when some residences and businesses overseed winter lawns and others do not. There was also great difficulty in distinguishing sparsely populated lawns and pervious surfaces, as most of the reflectance was from the soil and not the grass. Since we were most interested in delineating woody vegetation coverage, we attempted to use other metrics to discriminate woody vegetation from grass. Dormant grass was adequately discriminated against woody vegetation. However, overseeded lawns were only properly classified 72% of the time was likely caused by similar reflectance values to woody vegetation. Dark colored asphalt was also misclassified as woody vegetation in 48% of the observations.

Additional discrimination analysis was conducted with the other classes (buildings, roads, and pervious surfaces). However, strong cross classifications occurred among these classes and discrimination was not acceptable. Similar problems have existed in other urban forest classification systems that included the near IR (e.g. Myeong 2001), and can be attributed to common substrate, and therefore a similar spectral response. It was also difficult to differentiate between dormant or sparsely covering grass and bare soil also due to similar spectral response (Stefanov et al. 2001).

This imagery was selected as a low-cost alternative to high-resolution multi-spectral aerial imagery, approximately 10% the cost of other multi-spectral imagery commonly used. The typical markets for true-color imagery are real estate and tax assessment, which require inexpensive, high-resolution color photography in order to entice and excise, respectively. This imagery is now being collected with 0.33 m resolution. Imagery is available for over 150 sites covering entire cities of varying sizes, and is photographed anywhere from 5-40 times per decade at the present rate, dependent primarily on the rate of development, making it an ideal target to follow urban growth.

The obvious disadvantage of using true-color imagery is the limited spectra available for analysis. Vegetation abundance is typically estimated by vegetation indices such as the normalized difference vegetation index (NDVI), computed as

$$\mathrm{NDVI} = \frac{(\mathrm{NIR} - \mathrm{Red})}{(\mathrm{NIR} + \mathrm{Red})}$$

NDVI is superior to many other vegetation indices due to the unique characteristic of vegetation to exhibit high reflectance in the near infrared band and a high absorbance in the visible spectrum by photosynthesizing mesophyll in the leaves (Sabins 1996). True-color photography lacks the NIR band, and thus cannot be used for establishing NDVI. However, we believe the broad availability of low cost, periodically collected, sub-meter resolution aerial photography necessitates adopting an alternative approach.

Conclusions

A classification scheme was presented that adequately classifies true-color aerial photography in order to highlight elements of the Phoenix metropolitan area’s urban forest. This classification scheme was parameterized in order to maximize the user’s accuracy of the resulting image. The intention of this goal was to produce a spatially explicit urban vegetation map that could be used to model ecological and environmental phenomena within the urban ecosystem. Certain elements of the urban forest were misclassified due to characteristics of vegetation including size and type. Producer’s accuracy was greatest for coarse, dense vegetation. Small vegetation, sparse canopy density, and gray foliage all confused the classification scheme. Other land cover types were improperly classified as woody vegetation, specifically dark asphalt and actively growing grass.

It is important to note that the transportability of this classification scheme should be applicable to imagery of similar quality in other arid cities. Certain parameters maybe transportable to cities in other biomes; however, this should be done with care. It is important to note one of the conveniences of remotely sensing desert cities; trees are not winter deciduous, typically. We capitalized on this phenomenon by analyzing imagery from a time in the year in which summer growing grass is dormant. If the caretakers do not overseed during the winter, these patches are quite different in spectral quality than woody vegetation, allowing for enhanced discrimination between these problematic land cover types.

TABLE 1. Segmentation parameters used for this analysis

Scale	Color	Shape
10	1	0

Scale is an index that defines the resolution of the succeeding image objects. Values for color and shape are weighting functions that range from 0 to 1. Segmentation was conducted at fine resolution, relative to the imagery, based only on spectral characteristics.

TABLE 2: Histogram overlap values for spectral metrics analyzed.

		METRIC			
		Mean	Variance	Ratio	Brightness
BAND	Red	0.05	0.37	0.72	*
	Green	0.06	0.32	0.09	*
	Blue	0.05	0.09	0.33	*
	All	*	*	*	0.11

Overlap values are for classes *WOODY* and *OTHER*. Ratio is defined as mean of band X / brightness, where brightness is the channel mean value for all layers

TABLE 3: General error matrix and accuracy statistics of classification scheme.

		REFERENCED AS		
		WOODY	*OTHER*	
CLASSIFIED AS	*WOODY*	702	29	731
	OTHER	222	392	604
		924	411	1335

Producer's Accuracy	User's Accuracy
WOODY = 0.76	WOODY = 0.96
OTHER = 0.92	OTHER = 0.63

Overall Accuracy = 0.81

$\hat{K} = 0.63$

TABLE 4: Producer's accuracy related to genus.

Genus	Omission Errors	N	Producer's Accuracy
Morus	0	19	1.00
Populus	0	10	1.00
Citrus	1	48	0.98
Ulmus	1	30	0.97
Fraxinus	1	18	0.94
Pinus	4	62	0.94
Eucalyptus	2	28	0.93
Ficus	3	30	0.90
Brachychiton	3	24	0.88
Prosopis	10	77	0.87
Nerium	8	61	0.87
Olea	3	19	0.84
Bougainvillea	3	17	0.82
Agave	2	10	0.80
Juniperus	2	10	0.80
Phoenix	5	18	0.72
Rhus	4	14	0.71
Parkinsonia	18	62	0.71
Acaica	8	24	0.67
Larrea	8	19	0.58
Leucophyllum	15	32	0.53
Caesalpina	8	15	0.47
Baccharis	6	11	0.45
Washingtonia	14	25	0.44
Fouqueira	10	16	0.38
Cassia	17	27	0.37

TABLE 5: Error matrix of *OTHER* referenced land cover types

Land cover	Ommission Errors	N	Producer’s
Grass			0.86
Dormant	0	73	1.00
Overseed	19	67	0.72
Gravel	0	73	1.00
Bare soil	0	115	1.00
Road			0.88
Concrete	0	57	1.00
Asphalt	10	26	0.62

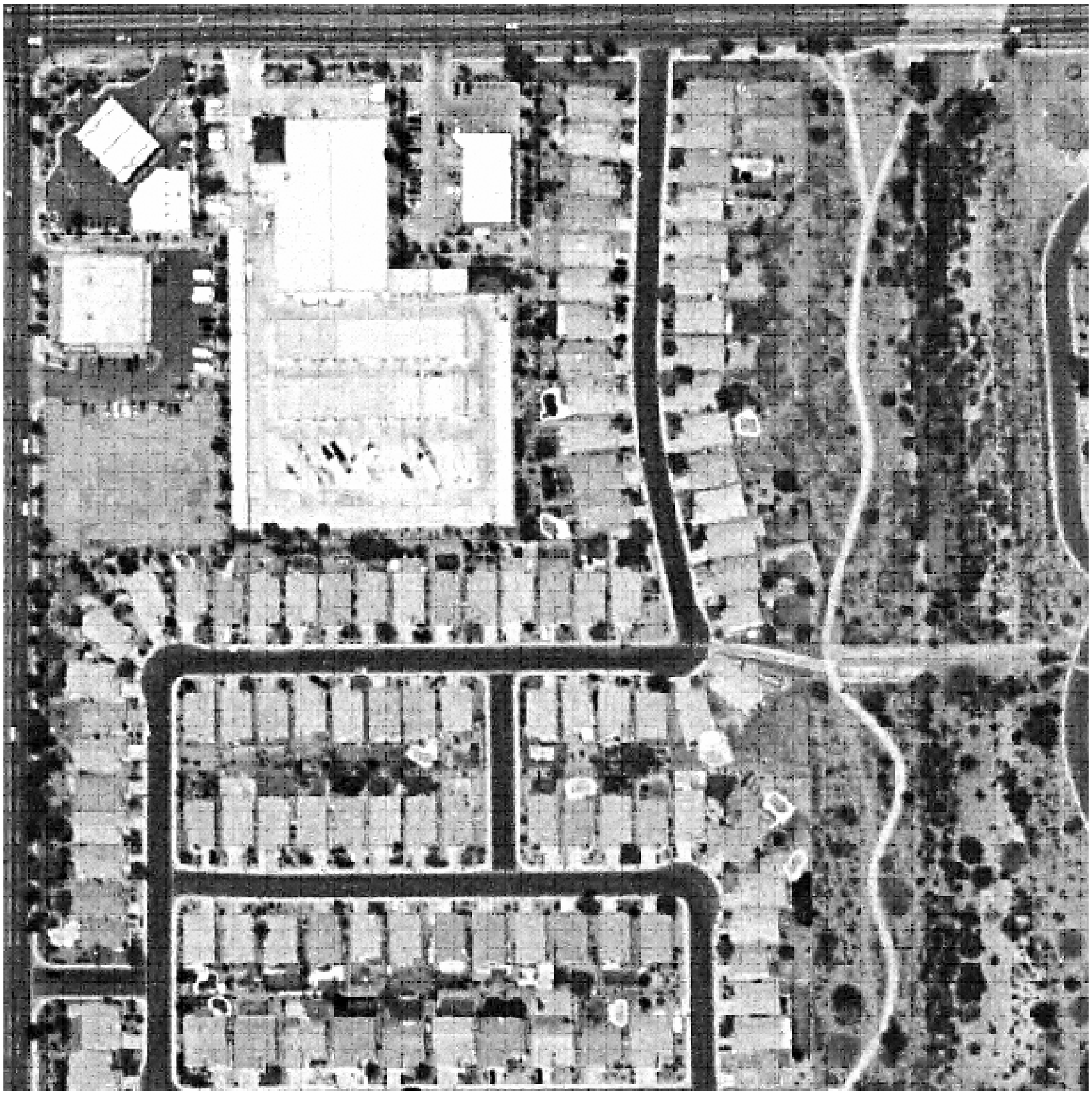

FIG. 6. Raw imagery (shown monochromatically), 0.61m spatial resolution.

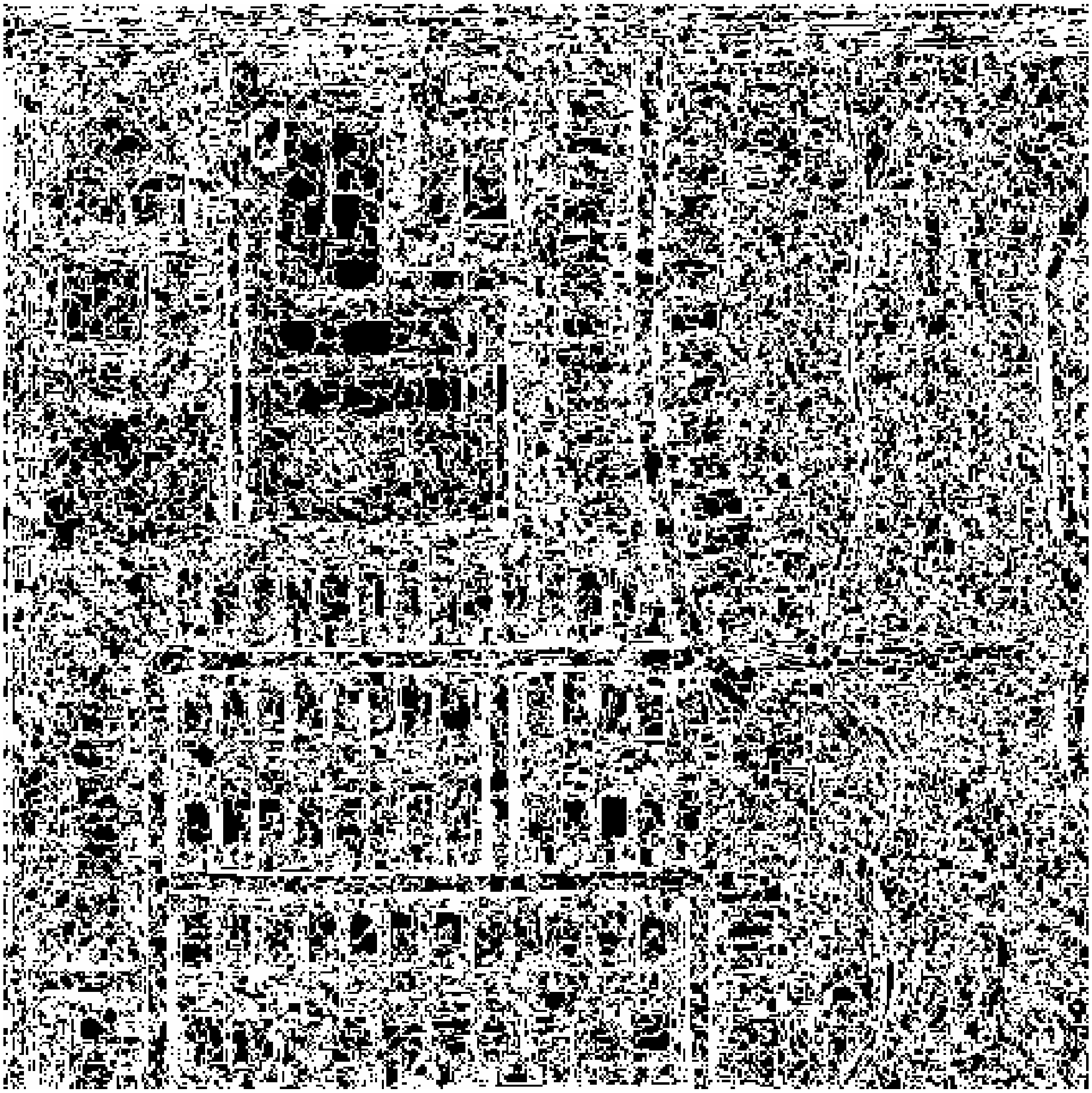

FIG. 7. Segmented image including raw image objects used for classification.

FIG. 8. Classified image. Vegetation in black.

FIG. 9. Woody vegetation of the Phoenix metropolitan area. Final classified image.

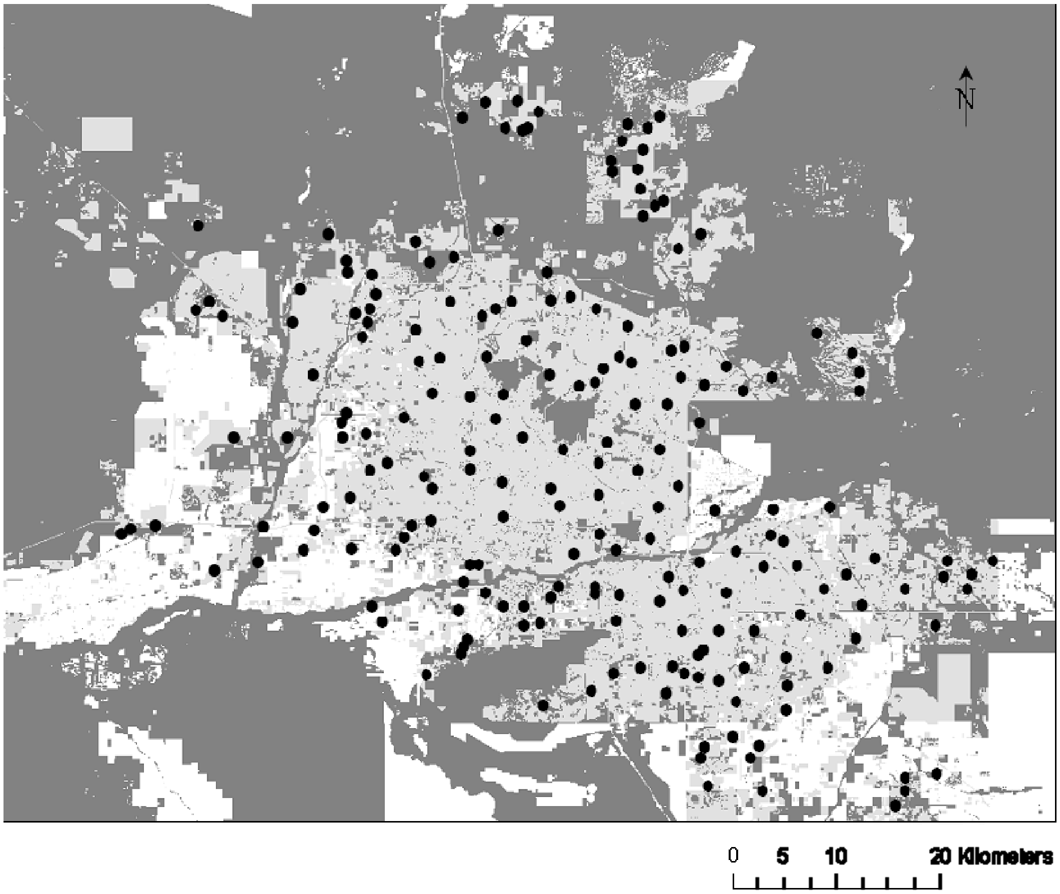

FIG. 10. Location of 203 randomly selected sites used in accuracy assessment, where light gray is urban, dark gray is remaining Sonoran Desert, and white is agricultural.

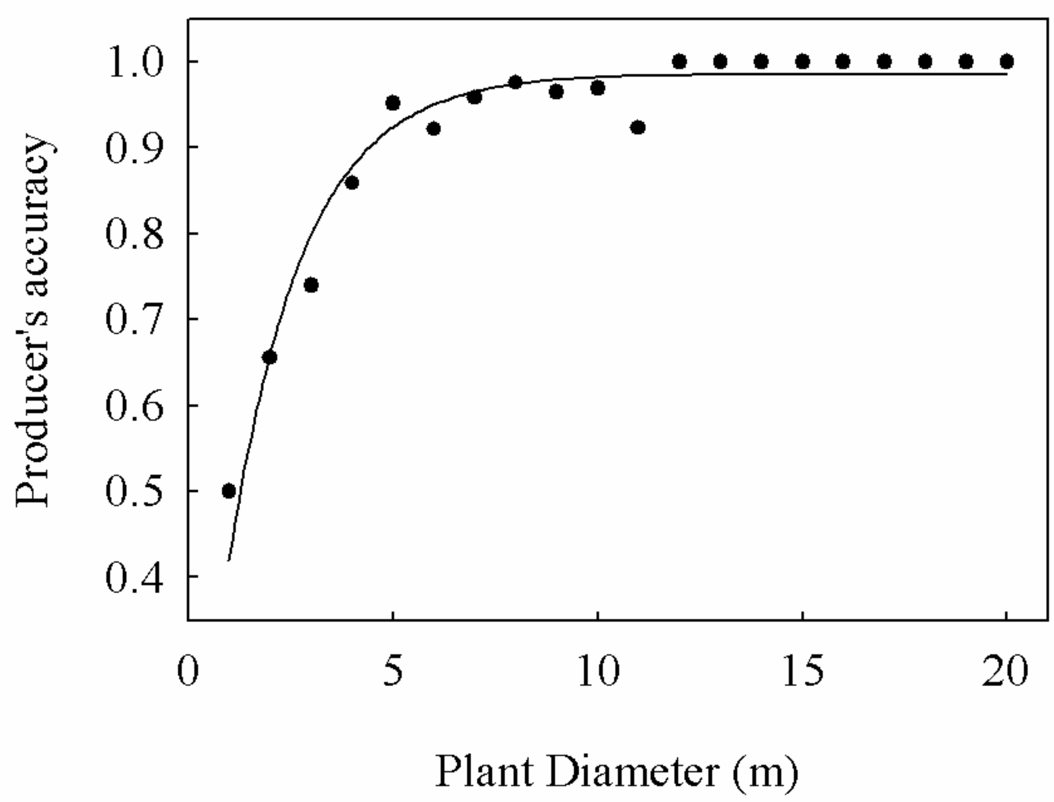

FIG. 11. Effect of plant size on omission errors. $r^2=0.94$. $p<0.0001$

CHAPTER 4

MONEY GROWS TREES: A SOCIO-ECOLOGICAL PATH ANALYSIS

This chapter examines the direct and indirect socioeconomic predictors of tree canopy cover in residential areas in Phoenix. The motivations for this focus are practical, theoretical, and methodological. From a practical perspective, trees have become the dominant biotic elements in nearly all urban settings in biomes ranging from deserts to grasslands and forests (Dwyer et al. 2000). Understanding the dynamics that determine urban vegetation patterns are important as plants provide numerous ecological and social services within cities. For instance, vegetation enhances air quality by absorbing particulate pollution (Randolph 2004), sequestering CO_2 via photosynthesis (Rowntree & Nowak 1991, Nowak 1994, Jenkins & Reimann 2003), reducing CO_2 emissions by decreasing ambient temperature (Rowntree & Nowak 1991), and by converting a reactive greenhouse gas, NO_3, into inert N_2 (Groffman et al. 2003). Urban vegetation enhances water quality by stabilizing urban streams and regulating their flows (Groffman et al. 2003), and reducing non-point water pollution (Randolph 2004). Urban vegetation ameliorates urban microclimates by reducing albedo and radiation (Sukopp & Werner 1982, McPherson et al. 1997), effectively enhancing energy conservation by reducing heating and air conditioning needs (Heisler 1986, Heisler et al. 1995, Gallo et al. 1993). Urban vegetation also contributes to human aesthetics (Wilson 1984, Morancho 2003) and enhances overall community well-being (Schroeder and Cannon1983) and psychological health (Ulrich 1984).

Human management has long been realized as an important determinant of urban vegetation dynamics (Sanders 1984, Sanders & Stevans 1984, Whitney & Adams 1980, Hope et al. 2003, Grove et al. 2006). However, vegetation management in urban ecosystems are determined by a broad spectrum of anthropogenic agents, from the decisions of individual homeowners to higher order effects such as land use change policies at municipal, regional, and federal levels (Chapter 2, Grove et al. 2005). Previous research focused on demographic and socioeconomic predictors of residential vegetation in urban areas has examined the extent and composition of vegetation structure. Many of these studies have examined the relationship between population density or social stratification and the extent or distribution of vegetation in urban ecological systems. For example, Iverson and Cook (2000) found that tree cover in Chicago, IL was negatively correlated with population density. Researchers found that socioeconomic status was an important predictor of plant species composition (Whitney and Adams 1980), diversity (Hope et al. 2003), and richness (Martin et al. 2004). Socioeconomic status has also been found to be significantly associated with vegetation distribution on private lands and public rights-of-way and potential space for vegetation planting (Grove et al. 2006).

A number of studies have also examined the utility of additional demographic characteristics associated with lifestyle behaviors such as household composition, ownership type, and residence time to predict the distribution of existing urban vegetation cover and urban vegetation structure (Grove et al. 2006). The inclusion of these lifestyle characteristics provided better results for predicting vegetation cover and structure on private lands than using population density and socioeconomic status alone (Grove et al. 2006).

The linkage between these lifestyle characteristics and urban vegetation is hypothesized to be associated with the social differentiation among urban neighborhoods that frequently becomes manifest in terms of lifestyle choices that households make and how those choices change over time. In the case of urban ecology, Grove and others (2004, 2006) have termed this phenomenon “An Ecology of Prestige,” referring to the phenomenon in which household patterns of consumption and expenditure on environmentally relevant goods and services are motivated by group identity and perceptions of social status associated with different lifestyles. In this case, a household’s land management decisions are influenced by its desire to uphold the prestige of its community and the lifestyle of the household (Grove et al. 2006).

A critical dimension that may be missing from the focus on population density, socioeconomic status, and lifestyle characteristics is a temporal component. A number of studies have shown that age of housing is significantly associated with plant species composition (Whitney and Adams 1980), diversity (Hope et al. 2003), and abundance (Martin et al. 2004). Moreover, researchers have found that age of housing is an important predictor for lawn fertilizer application levels (Law et al. 2004), distribution of vegetation and patterns of vegetation and potentials for greening (Grove et al. 2006). Thus, the combination of socioeconomic characteristics and housing age may improve the capacity to predict variations in canopy cover on residential lands.

Baltimore has been the focus of much of the recent research addressing the relationship between demographic and socioeconomic predictors of urban vegetation pattern (Grove et al. 2006). To date, however, this research has not been tested in the

American Southwest where in cities such as Phoenix, the climate is hotter and drier, and the housing stock is newer and development patterns are dominated by clustered, detached, single story homes.

Furthermore, previous research has not addressed the network of interactions that may potentially exist within socio-ecological data. Multiple regression is a common multivariate technique that attempts to maximize the variance of the dependent variable. However, it works under the assumption that all variables are independent, and consequently uncorrelated, which is rarely the case for social and ecological variables. One method to avoid this problem is to use principal components regressions which generate axes from the original independent variables that, by definition, are entirely uncorrelated. These axes can subsequently be regressed against the independent variable and not violate the assumption of independence. While this procedure can yield fruitful results, it does have problems. If many of the independent variables load highly on a single generated axis, it is difficult to deconstruct which variable is affecting the dependent variable, conflating interpretation. Furthermore, it is impossible to ascertain the relationships among those correlated variables. For this reason, social scientists have long used an alternative method, structural equation modeling (SEM), to study interacting multivariate hypotheses within a theoretical framework (Bollen 1989). This procedure is increasingly being adopted in the ecological sciences (Pugesek et al. 2003, Grace and Keely 2006). Path analysis is a subset of SEM that analyzes measured variables in order to test the predetermined theoretical effects of dependent variables on each other and their direct and indirect effects on one or more independent variables. This procedure provides means for statistical control of interdependency among the data within an *a priori* theoretical framework. Controlling such interdependency is important in nearly all socio-ecological systems due to the large degree of interconnectedness within social and ecological phenomena. In this chapter, path analysis is utilized in order to investigate the extent and distribution of vegetation within residential neighborhoods of the Phoenix metropolitan area and the effects that income and housing developmental patterns have on the distribution of vegetation.

Based on the focus, motivations, and challenges we have described, we present and test the following model in order to ask, "*What is the relationship between economic status and residential housing patterns in the Phoenix metropolitan area, and how do these interacting factors relate to distribution of canopy cover in urban residential areas*?"

A priori socio-ecological model

Path analysis is directed by an investigator's theoretical framework and knowledge of the expected mechanisms operating within that system. This analysis is formally presented as an *a priori* model (Figure 12). We hypothesized that canopy cover in residential neighborhoods in Phoenix would be *directly* affected by four social factors:

H_1: *Residential canopy cover will be positively correlated with household income* (Figure 12a). More affluent households are more able to install and maintain trees.

H_2: *Residential canopy cover will be positively correlated with home ownership* (Figure 12b). Enhanced tree stewardship occurs within neighborhoods of higher ownership.

H_3: *Residential canopy cover increases with neighborhood age* (Figure 12c). Phoenix has been mostly constructed within the last half century; therefore, canopy cover should be a positive linear function with neighborhood age.

H_4: *Residential canopy cover increases with enhanced urban sprawl* (Figure 12d). Increased densification constrains available growing space and therefore should limit the amount of vegetation that can grow.

To understand the effects of socioeconomics and housing variables on vegetation coverage, the indirect effects of socioeconomics on housing variables must be incorporated in the model. We test the following hypotheses:

H_5: *Economic status will be negatively correlated with housing density* (Figure 1e). As economic status increases, housing density decreases.

H_6: *Property ownership will be negatively correlated with housing density* (Figure 12f). As homeownership increases, housing density decreases.

H_7: *Housing age is negatively correlated with housing density* (Figure 12g). Newer houses are more densely developed, thereby reducing urban sprawl. As Phoenix continues to grow, space has become limited and therefore more expensive, causing developers to build more densely now than in previous decades (Gober 2005). Increasing housing density is also conflated by the market driving developers to construct larger houses.

H_8: *Economic status is negatively correlated with housing age.* More affluent residents will buy newer houses (Figure 12h), as predicted by Muth's (1969) "housing filtering model".

H_9: *Economic status is positively correlated with home ownership.* Higher income households are more likely to be able to afford home ownership (Figure 12i).

Methods

Study Area and Sampling Design

During the latter half of the twentieth century, the Phoenix metropolitan area of central Arizona has experienced exponential growth driven by its favorable climate, affordable housing, paradoxically cheap water, availability of jobs, and the commercial success of the air conditioner (Luckingham 1989). With over four million inhabitants, Phoenix has become the fifth largest and the fastest growing metropolitan area in the nation (US Census Bureau 2000). The Central Arizona-Phoenix Long-Term Ecological Research (CAP LTER) program was established to study the ecological impacts of this rapid urbanization. Ecological surveys are conducted every five years at 204 sites covering 6400 km^2 encompassing the entire Phoenix metropolitan area, its adjacent agricultural lands, and undeveloped upland Sonoran Desert. Permanent CAP LTER site locations were identified by an urban to rural dual-density, randomized tessellation-stratified sampling regime (but see Hope et al. 2003).

For this study, we used three spatially-explicit data sets to subset from these permanent plots (n=204) all the plots in which a single sample was identified where the sampling extent was (1) classified as residential by the Maricopa Association of Governments (2004), (2) within a census-defined block group (US Census Bureau, 2000), and (3) containing a CAP LTER survey point (n=112) (Figure 13). This approach was conducted in order to bound the extent of each sample in the analysis to correspond to the residential neighborhoods within a block group that precisely match the extent in which the US Census Bureau gathered economic and housing data. In other words, we only applied the urban forest classification to the precise extent that data on economics and housing variables were collected by the US Census Bureau, by only calculating vegetation cover within residential neighborhoods for each census-defined block group (n=112). Vegetation cover was calculated as described in Chapter 3.

Socioeconomic and Housing Data

The following socioeconomic and housing variables hypothesized to have an effect on vegetation coverage were selected for the appropriate block groups from the U.S. Census Bureau (2000): income per capita (USD), median house value (USD), number of housing structures, housing ownership (%), median year house built. Housing density (#/km^2) was calculated as the number of housing structures within the residential area as defined by MAG (2004), within the census-defined block group. Median year house built was used as an inverse proxy for neighborhood age, as most neighborhoods within the Phoenix metropolitan area have been created wholesale by developers in a more or less single time step (Gober 1998). Accordingly we created a variable, neighborhood age, calculated as the difference between 2000 (the year of the census) and the median year house built.

Path Analysis

Once a conceptual model has been created, the next step in path analysis is to relate the observed variables to their corresponding hypothesized statistical relationships in the form of a path model. We tested the initial path model, along with a series of alternative models, in order to assess the validity of our individual hypotheses. Prior to path analysis, all variables were examined for inappropriate distributional relationships and adherences to linearity. Raw data of all variables indicated significant deviations in normality. Proportional data (canopy cover & housing ownership) were transformed via an arcsine square root function, which is commonly used to normalize artificially truncated data (i.e. proportional data). In order to determine appropriate transformations for the remaining, non-proportional indicators, the Box-Cox procedure was conducted on the socioeconomic and housing variables using canopy cover as a dependent variable (Box and Cox 1964). This procedure suggested that logarithmic transformations would be appropriate for the remaining variables: income per capita, housing value and year built. Analysis of Lilliefor's test for normality (Lilliefor 1969) on the transformed variables indicated that the error terms of the transformed variables did not significantly deviate from normality and were observed to be linear. These transformed variables were used in the subsequent path analysis.

Model estimation was based on maximum likelihood. The suitability of model fit was assessed by analyzing the model chi-square and its associated *P* value (Table 6). It is important to note that insignificant *P* values represent no discrepancies between the model and the data, indicating an empirically appropriate model. This result is contrary to the interpretation of the results of more common statistical methods, such as regression and ANOVA. Additional fit indices (e.g. Aikike's Information Criterion, Bonferroni's Information Criterion) were also analyzed and indicated similar results to the chi-squared test. Individual path coefficients, representing the specific hypotheses, were tested via *z* tests through an iterative path deletion process in which the consequences of the deletion to the model chi-square were ascertained. Backwards pruning was conducted by successively eliminating the individual paths from the endogenous variable (the first of which was tree cover), assessing the model fit, and then re-entering the path, removing the next path, and so on (Table 6). The final model from this process was judged to be the best representation of the interactions among data, based on the comparisons of the objective measures of fit of the successive models.

Results

The base model developed from the conceptual model yielded a good fit with a chi-squared of 3.47 with a single degree of freedom resulting in a non-significant *P* value. Note that in path analysis, non-significant values indicate no significant discrepancy between the data and the model, and therefore an acceptable model. From the base model, we singularly eliminated the path from each of the exogenous variables to canopy cover, and reran the analysis. That variable was then re-inserted to the model and the next eliminated and tested, and so on. The elimination of path b, corresponding to H_2, enhanced the overall model fit, $P = 0.143$ compared with $P = 0.063$. In addition, the coefficient of path b in the base model was not significant ($P = 0.26$), supporting its removal. Elimination of paths c and d did not improve model fit and thus were retained. Removal of path b from the base model resulted in Revised Model A. A similar strategy was employed to analyze the suitability of paths e through g. These removals were not warranted. Therefore, we rejected the base model in favor of Revised Model A, hereafter "the accepted model".

The results from the accepted model are presented in Figure 14. A number of parameters are presented including (1) the total proportion of variation in canopy cover explained by model ($R^2 = 0.27$), and (2) the partial regression path coefficients standardized by the standard deviations. The path coefficients represent the predicted linear change of the endogenous variable, y, to the exogenous, x, in standard deviation units. The model results reveal that the extent and distribution of canopy cover (1) was both significantly and positively associated with income and neighborhood age as predicted by H_1 and H_3, respectively. However, the extent and distribution of canopy cover was not significantly correlated with ownership as predicted by H_2. Surprisingly the extent and distribution of canopy cover was significantly correlated with housing density, but the relationship was positive, not negative (H_4). In other words, canopy cover increased as housing density increased. Relationships among socio-economic and housing data were consistent with the original hypotheses (H_5-H_9). More affluent neighborhoods have higher ownership rates, and live in newer, denser communities.

Discussion

In Baltimore, Maryland, Grove et al. (2006) found that socioeconomic status was an important predictor of existing canopy cover and possible canopy cover in Baltimore, Maryland. This chapter provides evidence to extend the importance of socioeconomics in predicting variations of residential canopy cover in southwestern cities in the United States, such as Phoenix, where the climate is hotter and drier, and where the housing stock is newer and development patterns are dominated by clustered, detached, single story homes. Using an SEM path analysis approach, we also show that income is a direct determinant of canopy cover and also affects canopy cover indirectly through neighborhood characteristics. The relative strength of the causal pathway from household economic status to canopy cover (Figure 12a) indicates that socioeconomic status is the dominant, singular determinant of residential canopy cover in our model. Removing the path in the model between socioeconomic status and extent of canopy cover (Table 6) allowed us to test whether socioeconomics had a direct effect on the extent of canopy cover, or whether the effect of household economic status was mediated by neighborhood characteristics, such as ownership, age, and housing density. Removal of this path greatly affected the fit of the data to the model, indicating that the path should be retained. This suggests that socioeconomic status has a direct, and the strongest, association with canopy cover in our model. In this light, we conducted a univariate regression on the untransformed variables of socioeconomic status and extent of canopy cover, in which income singularly explained 33% of the variance in canopy cover (Figure 15). While this correlation is stronger than the final path model (Figure 14), we feel that the accepted model more accurately represents indirect and direct factors associated with the extent and distribution of canopy cover in residential areas that the univariate approach oversimplifies.

The consequence of urban densification in Phoenix on vegetation abundance is peculiar. The denser houses are constructed within neighborhoods the more vegetation cover exists. This is contrary to our hypothesis that the amount of vegetation would be constrained by available growing space, which is inversely related to residential development intensity. And it is contrary to analyses conducted in Baltimore which suggested that housing density is significantly and negatively associated with canopy cover (Grove et al. 2006). One explanation, the "thermal comfort hypothesis", involves the interconnectedness of economic and housing variables and the desire for reprieve from temperature extremes. In Phoenix, the data from this analysis indicate that more affluent citizens are living in newer houses in denser neighborhoods. These citizens have the ability to landscape their yard according to their own aesthetic; whereas less affluent families may not be able to landscape how they wish due to a constrained budget. In hot and dry climates such as Phoenix, the shade afforded by dense vegetation is a welcome reprieve from the summer heat. In a recent social survey of 806 Phoenicians, 96% responded that shade is a desirable commodity within their homestead (Harlan et al. unpublished data). So it is not surprising that people who have the means to ameliorate their microclimate by promoting vegetation abundance would do so, even though their newer neighborhoods tend to be more densely situated. A second explanation, the "three tree hypothesis", suggests that for every residence constructed, the developer installs a certain number of trees regardless of how large the lot is. Thus, as lot sizes decrease over time due to increasing housing demand and land expenses, and if developers are maintaining the status quo by installing a constant number of trees per residence, total tree density (and subsequently canopy cover) would increase even though housing density is also increasing. These potential mechanisms represent both bottom-up household (temporal comfort) and top-down (three-tree) anthropogenic effects, which could possibly be working in tandem to generate this result. Further work should address these hypotheses.

A second unexpected result was that there was no direct association between ownership and extent of canopy cover as predicted, suggesting no direct link between stewardship and vegetation abundance. However, the removal of the ownership variable entirely from the model indicated a very poor fit between the data and the model ($P < 0.001$). This suggests that there is an important indirect association between ownership and canopy cover. This indirect effect may be associated with a neighborhood characteristic not included in the model: neighborhood associations. Neighborhood associations are ubiquitous in the Phoenix area, controlling among many things what can be planted and how it is to be maintained. Currently, data on neighborhood associations' rules (covenants), practices, and levels of activity are not available. Indeed, they are closely guarded. If these types of data were legally available, they might help to explain this unexpected result.

As predicted, there was a positive and significant relationship between neighborhood age and canopy cover in Phoenix. This relationship may be a historical legacy of the metropolitan area's recent and rapid development. Phoenix is a young, ever-expanding city largely built within the last half century (Luckingham 1989, Gober 1998). As a result, we see a temporal lag of urban forest development. Recent research has found that the distribution of residential canopy cover is positively associated with canopy cover until about year 45, and then is negatively associated with housing age (Grove et al. 2006).

The analysis in this research was constrained by the overall long term sample (n=204) of the Central Arizona Phoenix LTER project and the subset of the sample to focus on residential areas (n=112). Increasing the sample size would benefit the analysis in several ways. First, it would increase the degrees of freedom in our analysis, thereby allowing inclusion of additional variables such as race, ethnicity, and demographic variables, such as family composition, that have been identified as important predictors of variation in urban residential tree canopy cover in Baltimore (Grove et al. 2006). Second, it would be possible to expose the analysis to a greater variety of ecological conditions that may be potentially affecting the existence of urban vegetation abundance, such as soil texture or nutrient availability. Finally, the net result of this approach would be to increase the predictive capacity of the subsequent, revised model.

Conclusions

This chapter demonstrates the importance of considering the influence of household and neighborhood characteristics and their interactive effects on the distribution of canopy cover in urban areas. In the sprawling cities of the American West, the majority of urban canopy cover can be found on private residential lands. Consequently much of the vegetation within Phoenix, and the social and ecological services it provides, may be associated with decision making processes of individual citizens at the household scale. Understanding the direct and indirect relationships between household socioeconomic and residential developmental characteristics are important to understanding the multi-faceted effects of urbanization on vegetation dynamics. Path analysis provides a unique technique to allow for the testing of complex conceptual models, which are inevitably inherent in any socio-ecological system. We have adopted this approach in order to test individual hypotheses within a complex model for an intensely urbanizing system.

Results suggest vegetation cover is directly affected by economic status, as well as indirectly through its affect on neighborhood characteristics, specifically neighborhood age and housing density. There was no clear direct effect of ownership of canopy cover; however, completely removing this variable from the model was not warranted suggesting important indirect effects. Interestingly, housing density was positively correlated with canopy cover, suggesting that more affluent people have on average more vegetation even though they exist in newer, denser neighborhoods. There was also a detectable temporal lag in vegetation abundance with neighborhood age, independent of the effects of income, ownership and housing density. This suggests that the effects of income are considerably important in the vegetation development of Phoenix, and that affluent communities have a disproportionately strong effect on the amount of urban vegetation within Phoenix. The drive for microclimate amelioration is a probable reason for this relationship. It is no secret that Phoenix is extremely hot in the summer. Thus enhancing the canopy cover becomes an attractive choice for ameliorating the temperature extremes within the urban environment, and it appears those with the means to enhance vegetation do just that.

TABLE 6: Path analysis decision tree.

Model	χ^2	df	*P*-value	*RMSEA*
Base Model	3.47	1	0.063	0.150
remove *a*	31.1	2	<0.001	0.364
remove *b*	3.89	2	0.143	0.093
remove *c*	6.80	2	0.033	0.148
remove *d*	6.63	2	0.036	0.148
Decision: retain paths *a*, *c*, & *d*; remove path *b*				
Revised Model A	3.89	2	0.143	0.093
remove *e*	10.9	3	0.012	0.155
remove *f*	13.9	3	0.003	0.182
remove *g*	10.5	3	0.015	0.151
Decision: retain paths *e*, *f*, & *g*				

Analysis of significance in path analysis differs from regression in that one seeks to produce a model that does not contradict the data, or statistically speaking, does not reject the null hypothesis. Backwards pruning was conducted by removing paths from baseline model beginning with those connected to the dependent variable (depicted in Figure 12) and assessing the improvement of the model. Deletion of path *b* from the base model was warranted suggested by an enhancement of the overall model's *P* value, suggesting that there is not a clear cut effect of ownership on vegetation abundance, as was predicted. All other paths were retained.

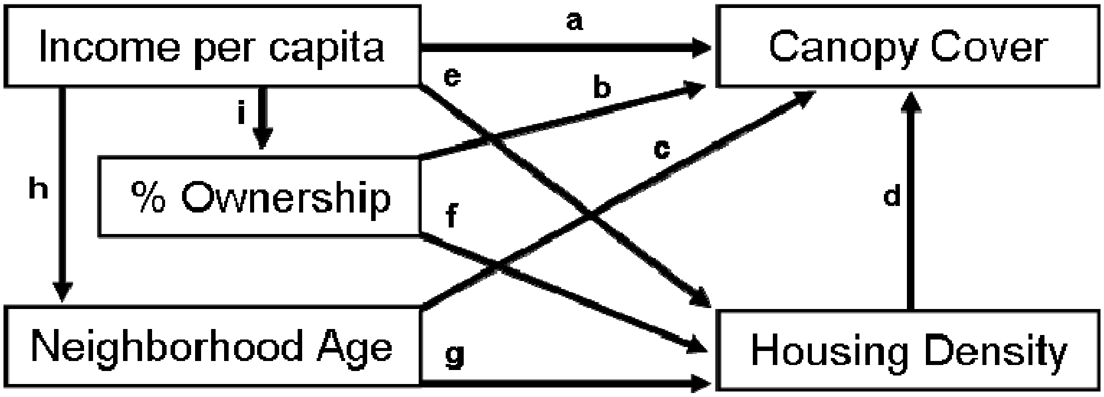

FIG. 12. The baseline conceptual model.

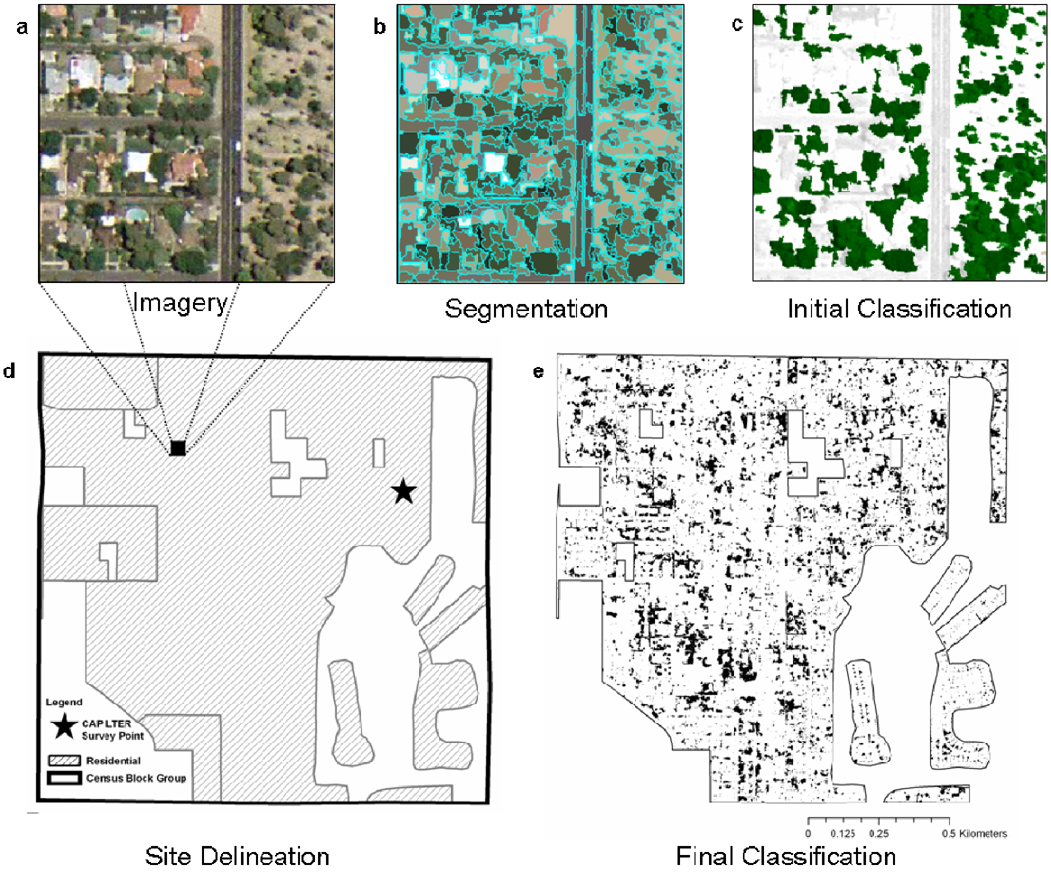

FIG. 13. The urban forest classification method.

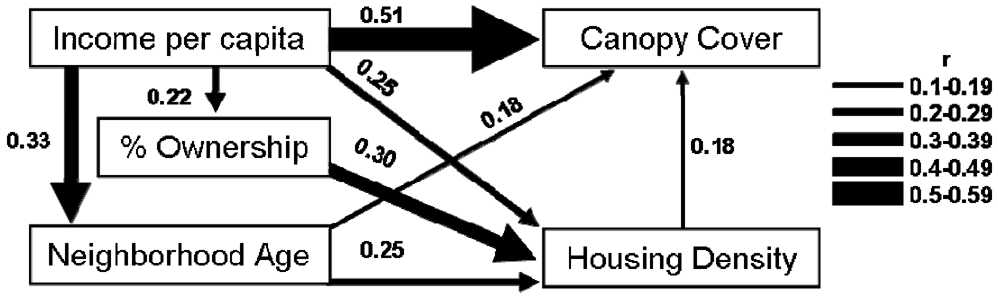

FIG. 14. Final model. χ^2=3.89 , df=2, *P* value=0.143, Model R^2=0.27. RMSEA=0.093. All paths are significant at α=0.05.

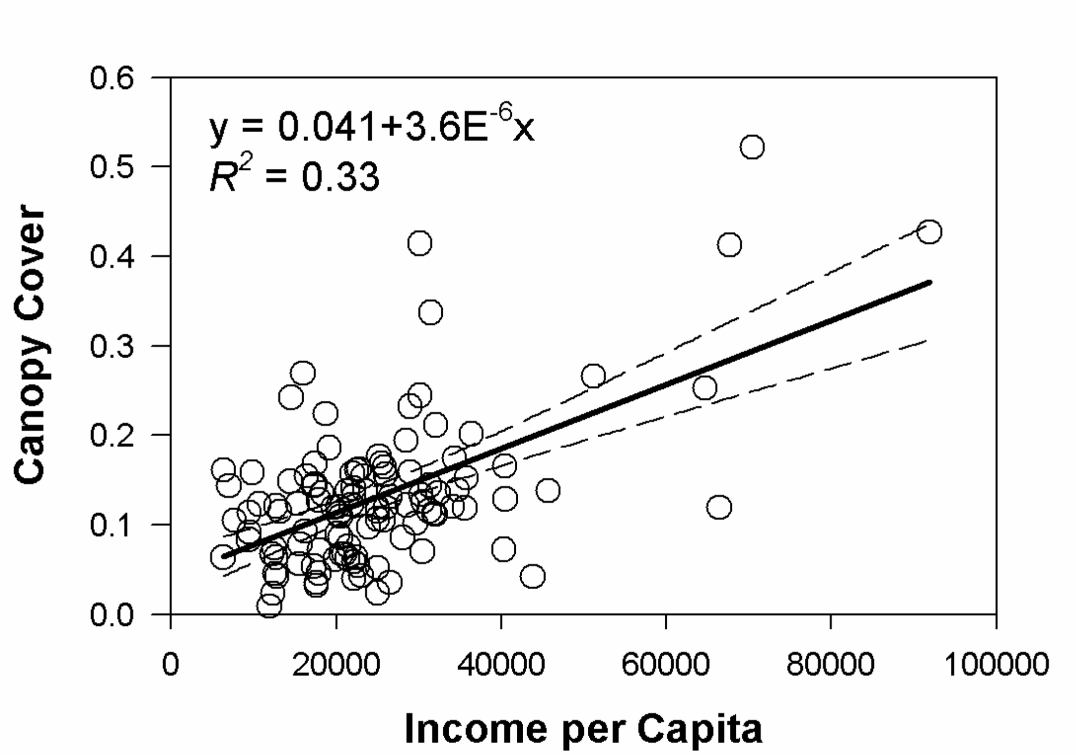

FIG. 15. Socioeconomics drive canopy cover. Simple regression with untransformed variables. Solid line represents the regression, whereas the dashed lines represent the 95% confidence interval. $p < 0.0001$

CHAPTER 5

EFFECTS OF LAND USE AND VEGETATION COVER ON BIRD COMMUNITIES

Urbanization is characterized by dramatic land use transformation, and is resultant of interactions among individual and political entities, which interact in tandem to impact ecological processes and landscape patterns from fine scales (e.g. homeowners) to broad scales (e.g. municipal developmental planning) (Hostetler 2001, Grove et al. 2006). Correspondingly, urban landscapes tend to be fragmented in a hierarchical manner resulting from these distinct anthropogenic scales which consequently affects animal habitat selection (Hostetler and Holling 2000). This chapter explores how the process of urbanization of the Phoenix metropolitan area affects the distribution of avian communities at two distinct scales. First, the effects of regional land use configuration on the bird community were analyzed by comparing the distributional patterns of various factions of the bird community with the spatial configuration of desert, agriculture and urban ecosystems. More locally, this chapter addresses the scalar effects between vegetation cover and the abundance of various avian functional groups, in order to determine which scales different types of birds respond to variations in vegetation cover.

Typically, urban areas harbor novel animal communities (for a review of novel communities see Hobbs et al. 2006), in which a few species increase in density relative to the surrounding areas, thereby creating distinct differences in community diversity between these two landscapes (Degraaf and Wentworth 1981, Blair 1996). This increase in avian density corresponds to a higher resource density within cities compared with wildlands (Chace and Walsh 2006), which is especially exaggeratedly in arid zones where water availability is scarce in the rural ecosystem (Shochat et al. 2004a). However within cities located in arid ecosystems, other resources may also play an important role in affecting bird abundance.

Land use transformation during urbanization consequently leads to land cover conversion, which can be a dominant process affecting ecological community structure and population dynamics (Hostetler 1997). Desert cities typically have elevated vegetation abundance compared with their rural counter parts (Dwyer et al. 2000, Chapter 2), which enhances both habitat for cover and nesting as well as food resources for birds. While the increase in vegetation and water are the main causes of the increase in bird population density, this increase is not typically followed by an increase in species diversity. In fact, most studies indicated lower bird diversity in urban areas than in adjacent wildlands (reviewed by Chace and Walsh 2006). Whereas wildland habitat degradation may cause loss of several native species, negative interactions between invasive and native species may be the major cause for the decrease in diversity in urban settings. Exploring the scale at which different guilds or groups of birds respond to vegetation will allow us to understand their constraints in rapidly changing environments, and serve as a starting point for understanding the mechanisms by which changes in community structure lead to loss of diversity in urban settings.

Animals utilize their environment across a range of scales, which is bounded by their extent, the broadest spatial area which organisms respond to their environment within their lifetime, and the spatial grain, the smallest area they respond to their environment (Kotlier and Wiens 1990). Within this range, organisms likely respond to their environment at a hierarchy of levels (Johnson 1980, Hutto 1985, Kotlier and Wiens 1990). Johnson (1980) recognizes four distinct levels of hierarchical habitat selection. At the very largest scale, *first order selection*, includes the entire area that an organism utilizes within its lifetime, and is also known as an organism's global home range or extent. In contrast, *second order selection* is an organism's local home range, or the area that it occupies within a unique ecosystem. This distinction is most apparent with migratory animals who utilize more than one distinct landscape for their survival (i.e. summer vs. winter feeding grounds), and much less so for organisms resident of one specific landscape for their entire life span. *Third order selection* is the selection of specific habitat patches within an ecosystem. For example, a Monarch butterfly would tend to select patches of milkweed within a prairie. And the lowest level, *fourth order selection*, involves the physical procurement of food within a selected patch, in our example, specific flowers within a milkweed patch, and is also known as grain.

Realizing the importance of hierarchical habitat selection, it has become apparent that single-scale studies of animals' responses to their environment may fail to adequately represent how that specific animal is responding to ecological parameter of interest, especially if they are not responding to the landscape at that scale (Holling 1992). The range of scales which an animal of interest is utilizing a landscape is important to determine prior to any further ecological investigation, as inappropriate scalar mismatch between organism and environment can lead to ambiguous or even deceptive conclusions (Wiens 1989, Levin 1992, Wiens et al. 1993).

To do this, we compared the correlation coefficients of bird abundances for different functional groups (e.g. foraging guilds, natives vs. exotics) with vegetation cover, as a proxy for habitat, across a range of scales (from 100m to 10km). Theoretically, a unimodal (hump-shaped) relationship should exist for the correlation coefficients across a range of scales, under the assumption that vegetation cover is an adequate estimate of bird abundance. The peak of that relationship, if statistically significant, would represent the strongest correlation between habitat and bird abundance, and thus signifies the *average* third order selection unit for that group. A strong peak is expected for species directly dependent on vegetation for food (herbivores), a weaker peak for omnivores, and the weakest relationship for those species indirectly dependent on vegetation (insectivores). The regional distributional patterns of the varying bird functional groups was also estimated by utilizing interpolation techniques designed for avian censuses in urban systems. Exotic species were expected to be spatially aligned to the urban ecosystem, and native species tied to the desert ecosystem. Herbivores were expected to exist in higher densities were vegetation is greatest, which typically exists within the city and agricultural fields in arid ecosystems (Dwyer et al. 2000).

Methods

Study Area and Sampling Design

We conducted this work using data collected as part of long-term ecological monitoring by CAP LTER, a research group studying the social and ecological effects of rapid urbanization on a Sonoran Desert ecosystem. The CAP LTER study area occupies 6,400 km^2 encompassing the Phoenix metropolitan area, agricultural lands and the outlying Sonoran Desert. The core monitoring effort of CAP LTER consists of a point sampling regime in which a suite of ecological variables are measured at a set of sites falling within the city, agricultural fields, and desert surrounding the Phoenix metropolitan area (see Chapter 2). Additionally, birds are monitored seasonally at 40 sites, randomly selected using a dual-density tessellation-stratified procedure, with randomly selected sites in the surrounding desert area occurring at one third of the density of sites in the urbanized areas (Fig. 16) (Walker et al. *in press*). This design maximizes the density of points within the highly heterogeneous urban landscape while maintaining a spatially balanced sample at the regional scale.

Bird Counts

Beginning fall 2002, seasonal bird counts have been conducted at these sites by three observers per site, including winter (January), spring migration (April), summer breeding (July), and fall migration (October). Thus, over a calendar year, each of the sites was visited 12 times (3 observers by 4 seasons). Observers employ 15 minute open-radius point counts, noting all birds and which species seen or heard and their estimated distance from the observer. To maximize numerical robustness for statistical analyses, we aggregated the counts of species across four seasons, occupying one calendar year (January-December 2003). We subset only data from 2003 only to correspond temporally with the imagery used for the vegetation analysis (see below). From the resultant community matrix, we calculated a suite of community-level variables for each site. First, the abundance matrix was split into two matrices, in which native and exotic bird abundances were calculated. Native species were those species historically extant in the Sonoran Desert as defined by Mills and others (1989). Second, abundance matrices were calculated for the following foraging guilds: granivorous, nectivorous, omnivorous, insectivorous, and predatory. Only two predators were sighted, and thus not analyzed due to the lack of numerical robustness.

Analyses

Avian Distribution Interpolation

Interpolation is a geostatistical method for predicting values between point data and visualizing those values through the creation of spatially explicit maps. Ordinary, simple, and universal kriging are not permissible interpolation techniques for data that do not conform to normality. Typically, this is not problematic if the data can be transformed to meet the assumptions of normality. However ecological censuses across multiple ecosystems characteristically produce heavily right-skewed data which does not lend itself to normalization via standard transformation techniques (Walker et al. *in press*). An alternative approach is to use indicator kriging with the untransformed data (Journel 1983). Indicator kriging does not assume normal distribution. Rather, it proceeds just like ordinary kriging, only for binary (indicator) variables. While the disadvantage of this approach is that specific estimates of populations cannot be made across the study area, it is a sufficient approach to map habitat distributional patterns (Walker et al. *in press*). Indicator kriging was conducted for each of the avian guilds (Fig. 18) and for native and exotic species (Fig. 19), in order to determine the unique pattern of the second-order (or first order for residents) selection range for those bird types. No distinction was made between first and second order selection in this analysis.

Differences were tested between the *measured* bird counts among the regional land uses for the foraging groups (Fig. 18e-h), native species (Fig. 19c), and exotic species (Fig. 19d). Differences were also tested between measured vegetation cover at the same scale in which the bird census data were collected (40m buffer) among the regional land uses (Fig. 17). Due to the non-normal nature of the data, overall significance was tested by the Kruskal-Wallis test for equality of factor levels (Kruskall & Wallis 1951), and multiple pairwise comparisons were conducted using Dunn’s method (Dunn 1964). For Kruskall-Wallis tests, df = 2. $\alpha = 0.05$ for all tests.

Vegetation Scalar Analysis

To conduct a scalar analysis at fine resolutions across a wide range of scales, an automated urban forest classification (as described in Chapter 3) was applied to a gradient of scales surrounding the bird count locations. Following classification, vegetation cover was calculated for a series of radial buffers from 100m to 10km in 100m increments. For each scalar increment, the proportion of canopy cover was determined and recorded for each site. These data were transformed via an arcsin square root function, customary to normalize artificially truncated proportional data. The transformed variables did not show significant deviations from a normal distribution. Correlation coefficients were calculated comparing the bird abundance of the various functional groups (e.g. foraging guilds, native vs. exotic) versus the proportion of canopy cover for each scalar increment, providing 101 correlation coefficients for each functional group. These correlation coefficients were plotted against the scale at which they were collected to determine whether there may be a distinct scalar response of vegetation cover to guild abundance (Fig. 20) and native vs. exotic abundance (Fig. 21) of birds.

Results

Interpolations of the bird counts provide visual representations of the spatial distribution of the different functional groups in a regional context. In concert with the non-spatial analyses, results indicate that the birds that directly depend on vegetation occur in higher abundances in the human ecosystem (e.g. urban and agricultural land uses) which characteristically has more vegetation than the outlying desert (Fig. 17). Dunn’s method suggests that nectivorous birds, all of which are native to the Sonoran Desert, and granivorous birds occur with greater frequency within the urban and agricultural areas than the outlying desert. Spatial interpolations of these foraging guilds supports this, but also indicates that nectivorous birds appear in the urban core with the greatest densities relative to other land uses (Fig. 18b); whereas granivorous birds are spatially located surrounding agricultural fields (Fig. 18a). Omnivorous birds, too, occur with greater frequency within the human ecosystem than the desert (Fig. 18g); however in contrast with the herbivores, they appear to spatially occur in agricultural fields with similar abundance to the city (Fig. 18c). In contrast to these patterns, birds that are not directly dependent upon vegetation, insectivores, exist with greater frequency in the outlying desert but also appear benefit from the existence of the agricultural fields (Fig. 4c,h)

Results from the vegetation scalar analysis (Fig. 20) indicate that those species that are directly dependent upon vegetation (granivores, nectivores, and omnivores) are significantly and positively correlated with vegetation across a wide range of scales. Furthermore, those species strictly dependent on vegetation (granivores and nectivores) exhibit a unimodal relationship with vegetation abundance and scale; whereas the omnivorous birds' relationship with vegetation abundance is significant and stable above 2000m, but lacks unimodality. In contrast, insectivorous birds are not significantly related to vegetation at any scale.

As expected, exotic species are highly bounded within the urban ecosystem and do not deviate far beyond (Fig. 19b,d). Surprisingly, native species appear to exist in equal abundances in both the city and the desert, and exist in highest abundances in agricultural fields (Fig. 19c). However when the spatial interpolation of the native birds are analyzed the highest (Fig. 19a; in the northeast and south central) and the lowest abundances of native birds (Fig. 19a: in the west and northwest) occur in desert sites. The areas where bird abundances are highest are unique in that they are agricultural fields adjacent to mountainous desert remnants with a unique high-diversity desert plant community, known as upland Sonoran Desert characterized by mountainous terrain. The area where the interpolated abundances are lowest is dominated by a low-diversity desert plant community, characterized by flat, desert scrub dominated by *Larrea tridentata*, creosote bush.

The relationship between the abundances of birds and vegetation of exotic vs. native species across scales shows striking juxtaposition (Fig. 21). At the lowest scale, native birds are significantly and positively correlated with vegetation, while exotic birds are not significantly correlated. When the window of observation is enlarged this pattern flips; such that native birds are only significantly correlated with vegetation abundance until a radial buffer of 250m. The correlation between vegetation and exotic birds is consistently significant beyond a 100m buffer. It does possess a unimodal relationship indicating that these species on average are most highly correlated with vegetation abundance at an approximate radial buffer of 2000m. However, this unimodality is weak.

Discussion

For most bird species in Phoenix, Hostetler and Knowles-Yanez (2003) showed that urban land use (e.g. residential, commercial, transportation) was not a good predictor of avian abundance. They suggested that this effect is likely due to a potentially low correlation between urban land use and the underlying land cover, such as vegetation. For example, tremendous variability in vegetation abundance can occur within residential neighborhoods. Established neighborhoods may consist of entire blocks that are heavily vegetated, providing sufficient canopy cover, food, and nesting locations; whereas newer residential developments are characteristically depauperate of standing biomass. Therefore, comparing avian responses to artificial delineations of the urban ecosystem (i.e. residential vs. commercial) may not be prudent.

While lower-order urban delineations of land use may not affect avian dynamics, we have shown that there are clear ecological consequences of land use on avian dynamics at a more regional scale when we compare avian distribution patterns of urban areas, more generally, with agricultural fields and the desert ecosystem. Native species are spatially present throughout the region, including the urban core. Analysis of this group highlights how considering the spatial distribution of a group in concert with traditional non-spatial analyses complement each other and enhance overall ecological knowledge. Non-spatial statistical analysis indicated that native bird abundance measured *in situ* was greatest for agricultural samples. This suggests that native species are able to utilize the enhanced productivity of this ecosystem to such an extent that they persist there in greater numbers than their endemic ecosystem. Surprisingly, native species also utilize the urban ecosystem in numbers that rival the desert, suggesting that urbanization may not be necessarily negatively affecting native bird *numbers*. However, changes in diversity of native species was not compared.

Consideration of the spatial distribution of the native birds leads to a different insight. The highest *interpolated* abundances of native species are found in areas where agricultural fields meet the mountainous desert ecosystem (in the S and NE corner of the study area; Fig. 19b). Such mountainous areas tend to have higher vegetation abundance and botanical diversity than desert lowlands (Chapter 2), which may contribute to higher bird abundance due to enhanced resource abundance and diversity. We interpret this pattern in concert with non-spatial results to indicate that native species occur in highest numbers where high-diversity desert meets high-resource farms. In this way native birds can access resources from their evolutionarily-adapted ecological community, and are able to supplement it with additional resources, most namely water. In contrast, the areas of lowest native bird abundances within the entire study area are creosote flats (in the NW corner of the study area; Fig. 19b), which typically have the lowest botanical and geophysical diversity within the desert (Chapter 2).

We found that native species also exist at relatively moderate abundances within the urban ecosystem relative to these extremes within the desert. This suggests that the urban ecosystem does not necessarily serve as a rigid ecological barrier for native species of birds. In fact it appears that the urban ecosystem is more advantageous for native birds than the low-diversity creosote flats. Further examination of the pattern of native bird distribution within the city indicates that the two areas of highest abundance (S and NE of region) are connected by an area of moderately high abundance within the city. The center of this pattern is present directly over the Salt River that bisects the city. Furthermore, the areas directly to the north and south of the Salt River are historically flood irrigated and characteristically have enhanced vegetation abundance. We interpret the interpolated pattern of native species distributions as an indication that these species may be utilizing the combination of these features as an ecological corridor to pass through, and even inhabit, the city. This pattern also exists for insectivorous birds, even more intensely (Fig. 18d).

Three of the four bird guilds (granivores, nectivores and omnivores) showed higher abundance in the urban and agricultural areas (Fig. 18a,b,c). This concurs with previous studies, reviewed by Chace and Walsh (2006), that as bird feeders supply food for these three guilds their densities are enhanced. Insectivores, however, are not associated with the urban habitat (Fig 19d). This result is intriguing. On one hand, bird feeders that normally enhance the amount of available food for birds do not include proper food for this guild in most cases. On the other hand, previous research from the CAP LTER study area indicates that arthropod abundance is higher in the urban habitat than in the desert (McIntyre et al. 2001, Shochat et al. 2004b, Faeth et al. 2005). Faeth and colleagues (2005) also showed a strong top-down effect from avian predation on urban ground arthropods from our sites. Our results may therefore suggest that top-down controls on arthropod abundance in the urban habitat are mostly caused by omnivorous birds. Possibly, species of this guild out-compete insectivorous species; or insectivorous species could be simply more sensitive to the change in habitat structure than other guilds. Future research should focus on the insectivore guild to address these hypotheses.

Our guild-based analysis of the scalar effect of bird abundance on vegetation produces two insights. First, the stronger the dependence on vegetation for food, the stronger correlation exists between guild abundance and vegetation cover (Fig. 20). In general, granivorous and nectivorous birds, those most directly dependent on vegetation, have the highest correlation with vegetation cover. While omnivorous birds are significantly correlated with vegetation at moderate to large scales, these correlations are consistently lower than the herbivorous species. This can be attributed to their incomplete dependence of vegetation as a direct food source. Insectivorous birds show no significant correlation between vegetation abundance. Second, the stronger the dependence in vegetation for food, the smaller the scale at which the peak correlation between vegetation and bird abundance occurs (Fig. 20). Granivorous birds appear to be most strongly correlated at relatively fine scales, indicating a third order selection range from approximately 500m to 2000m. Considering their tendency to occupy agricultural areas within this ecosystem (compare Fig. 16 with Fig. 18a), this pattern suggests granivorous birds are benefiting from the concentrated resources that are contained within relatively small areas characteristic of these areas. Thus, granivorous birds have a relatively small foraging range likely supported by concentrated resources of seeds within agricultural fields. While nectivorous birds also possess a unimodal relationship with scale, it is weak and the peak of this relationship is at a coarser scale relative to the granivorous birds (Fig. 20). This indicates that these species likely have a third order selection range between approximately 1500m and 5000m. The unimodal relationships between the abundances of the granivores and nectivores with vegetation across scales suggest that these groups tend to be restricted to unique scales.

In contrast, omnivorous birds are not significantly correlated with vegetation abundance at fine scales, but are significantly correlated at moderate to coarse scales. There does not appear to be strong unimodal relationship between omnivorous bird abundance and vegetation abundance indicating that these species likely have a broad spectrum of spatial responses to vegetation abundance, with a third order selection range from approximately 2000m to at least 10km. Interestingly, insectivorous birds do not have a significant relationship to vegetation at any scale. Although some species of this group use vegetation to perch or find insects in; others may feed in open areas away from vegetation (e.g. swifts, swallows, martins), or perch on artificial structures (e.g. antennas, poles, buildings). We speculate that this high variation in life history traits, combined with the weaker dependence of this group on vegetation, account for the lack of correlation between vegetation and bird abundance in this case.

The opposing response of native and exotic birds to vegetation is even more intriguing. Native birds respond to vegetation at the smallest spatial scale, with no correlation between vegetation and bird abundance beyond 250m. Exotic species show a broad range of spatial scales in which the correlation is relatively high and significant.

Interestingly, the correlation is low at smaller scale but increases, to reach the maximum at around 2000 m before decreasing again. We believe that this pattern indicates an important difference in the way these two groups utilize their environment. Vegetation abundance in the Sonoran Desert is highly variable. Riparian corridors can harbor completely closed canopies; whereas the upland environments may be completely devoid of vegetation. In desert ecosystems, this variability is largely controlled by available water (Chapter 2), which concomitantly influences the amount of food and nesting locations available for birds. In contrast, exotic species appear to more or less equally occupy the urban core (Fig. 19b) and are significantly related to vegetation above 100m (Fig. 21). Resource availability in the urban ecosystem is spatially and temporally homogeneous relative to the surrounding ecosystem (Shochat 2004, Anderies et al. 2007). The significant correlation between exotic abundance and canopy cover across scales may reflect this phenomenon, as exotic species can find food and water away from vegetation (e.g. artificial fountains, bird feeders, and refuses dumps).

Conclusions

How anthropogenic factors affect animal abundance and diversity in urban settings remains an important question in the ecological sciences and has clear consequences for conservation management as our world becomes more urbanized. The findings from this analysis suggest that data on habitat configuration within urban ecosystems should be collected around census sites should be collected at approximately a 2,000 m radial buffer (or approximately a 12.5 km^2 area) surrounding the site. This sample size tends to encompass most of the significant relationships, with the exception of native species, generally, which are better analyzed at much finer (<250m buffer) scales. We found that granivorous birds prefer urban and agricultural lands at a regional scale and are spatially correlated with vegetation at relatively fine scales. Nectivorous birds, all native to the Sonoran Desert, have become urban specialists and appear to be spatially correlated with vegetation at relatively moderate scales. Regionally, omnivorous birds are the most broadly dispersed guild and are positively correlated with vegetation from moderate to large scales but lack a distinct peak scale. Insectivorous birds are spatially confined to the desert and are not correlated with vegetation structure. Exotic bird species are urban specialists and are correlated with vegetation from moderate to coarse scales. Native bird species occupy regions with greatest abundance where agricultural lands meet the botanically diverse, mountainous Sonoran Desert. Native birds also appear to utilize the urban ecosystem with greater abundance than the low botanical diversity creosote flats of the Sonoran Desert.

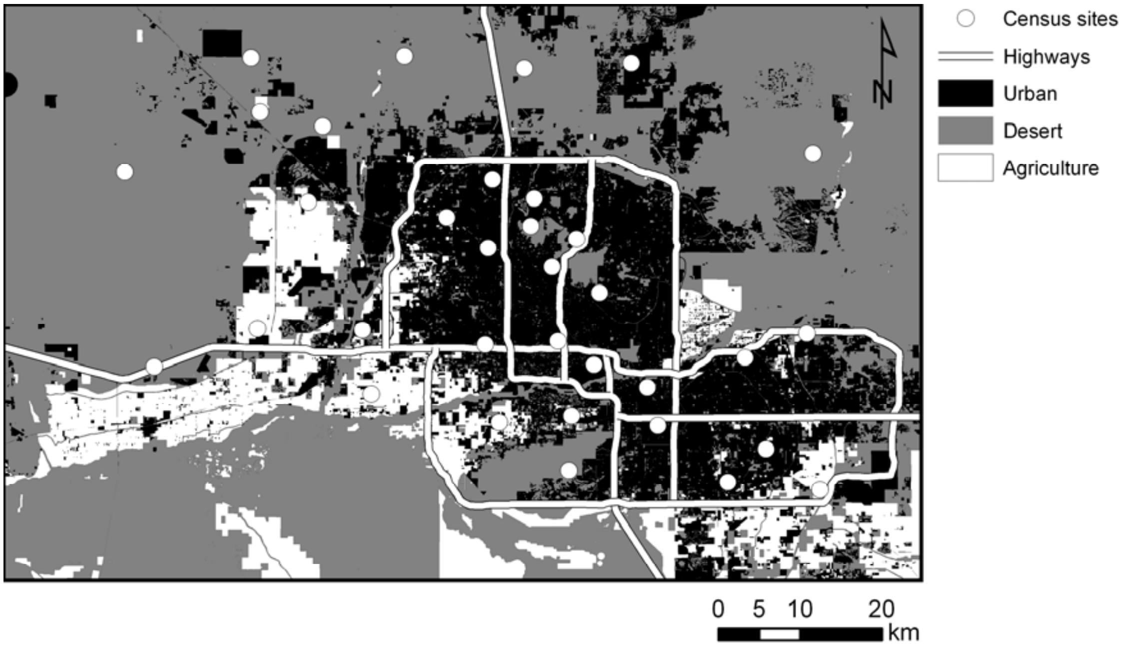

FIG. 16. Avian census locations and land use map of central Arizona (USA) incorporating the Phoenix metropolitan area.

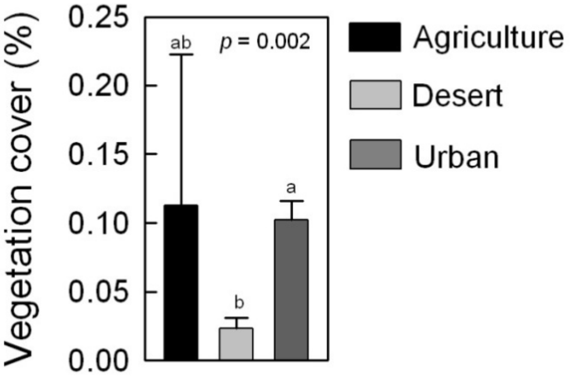

FIG. 17. Measured vegetation cover at the local scale, a 40m buffer surrounding the bird census sites. Kruskal-Wallis test was used differences between the local measured vegetation coverage. Letters refer to factors that were different according to Dunn’s test for pairwise comparisons. α=0.05 for all tests. Bars refer to mean + SE.

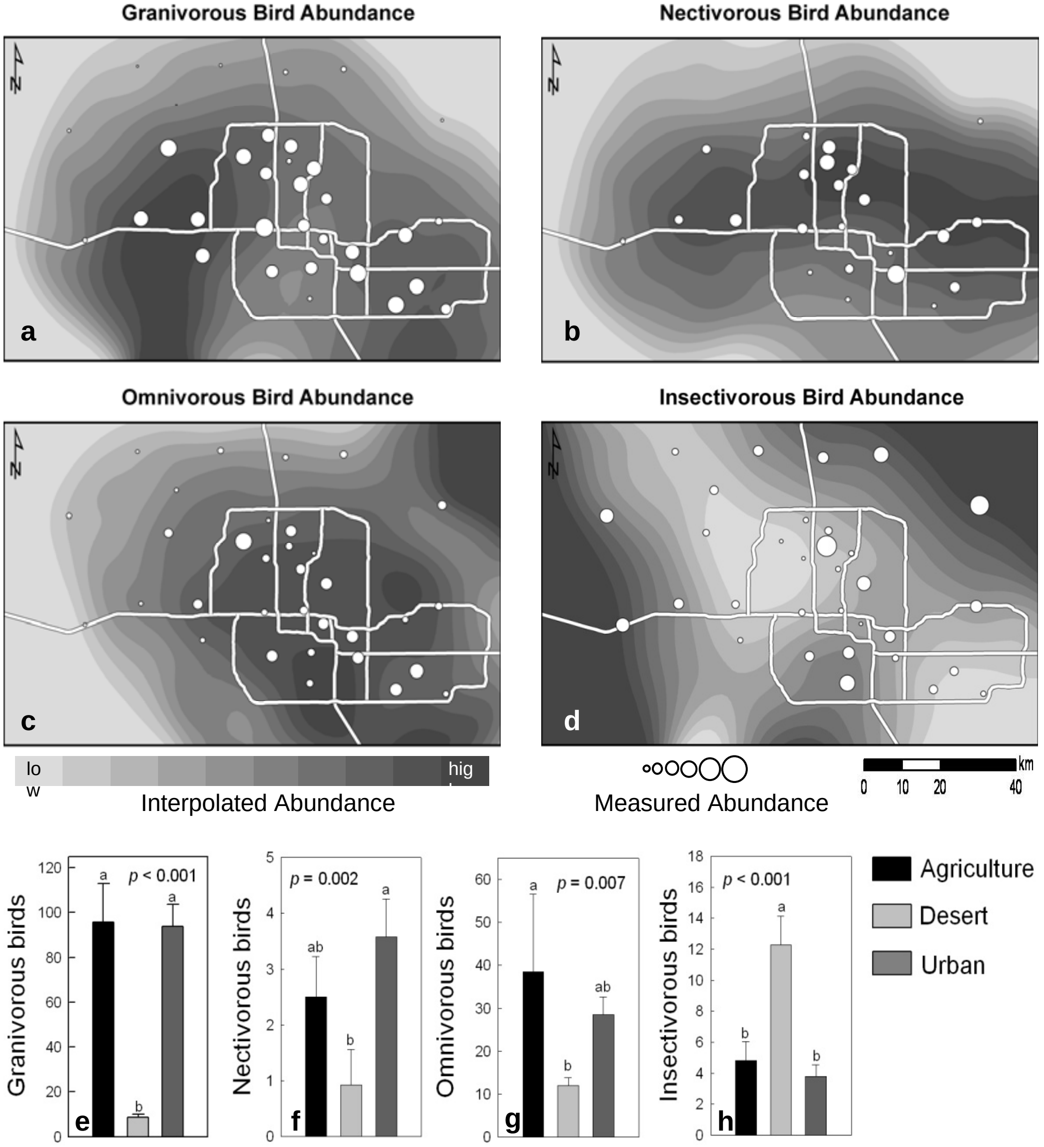

FIG. 18. Measured guild abundance and interpolated habitat distributions. Granivorous birds strongly prefer agricultural fields (a); nectivorous and omnivorous birds are urban specialists (b, c); and insectivorous birds are almost exclusively within the outlying desert (d). Interpolated extents are equivalent to Figure 16. Kruskal-Wallis test was used differences between the measured bird counts. Letters refer to factors that were different

according to Dunn’s test for pairwise comparisons. α=0.05 for all tests. Bars refer to mean + SE.

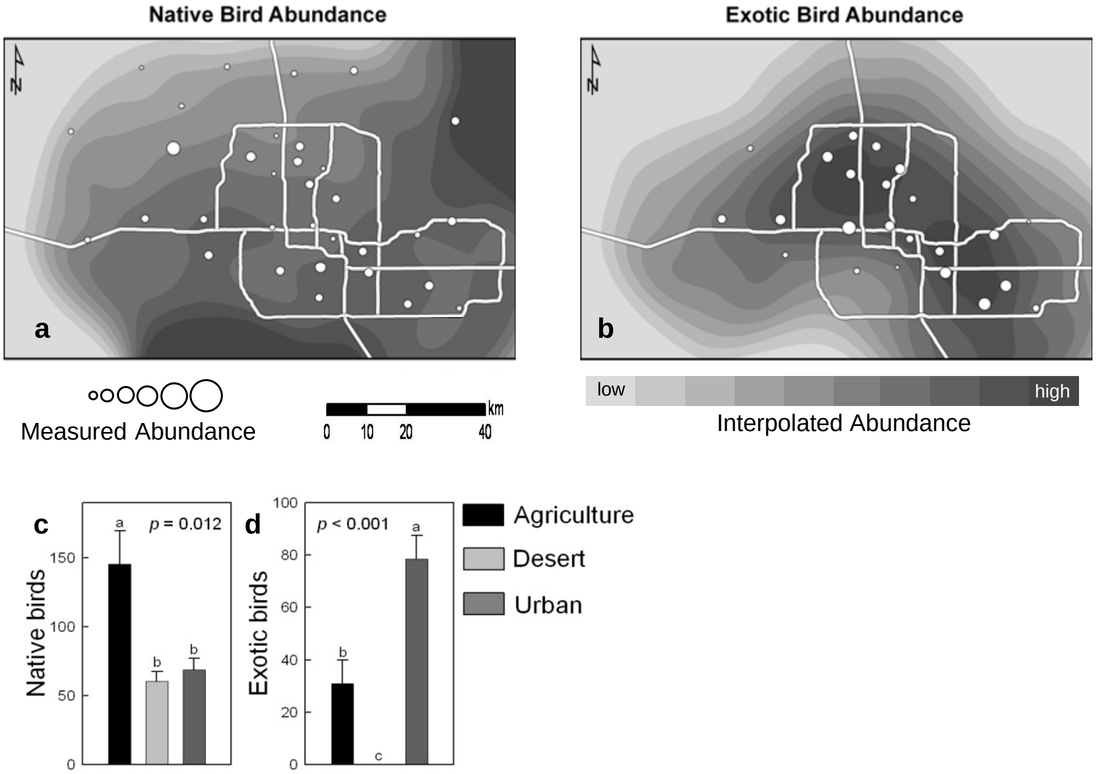

FIG. 19. Measured abundances and interpolated habitat distributions of native (a, c) and exotic (b, d) species. While exotic species are isolated to the urban core (b, d), native species utilize agricultural fields with the greatest abundance and with equal abundances in both the desert and the city (c). However, the spatial interpolations suggest that the highest densities of native species occur in mountainous regions of the desert (a). Interpolated extents are equivalent to Figure 16. Statistical tests were conducted as in Figure 18.

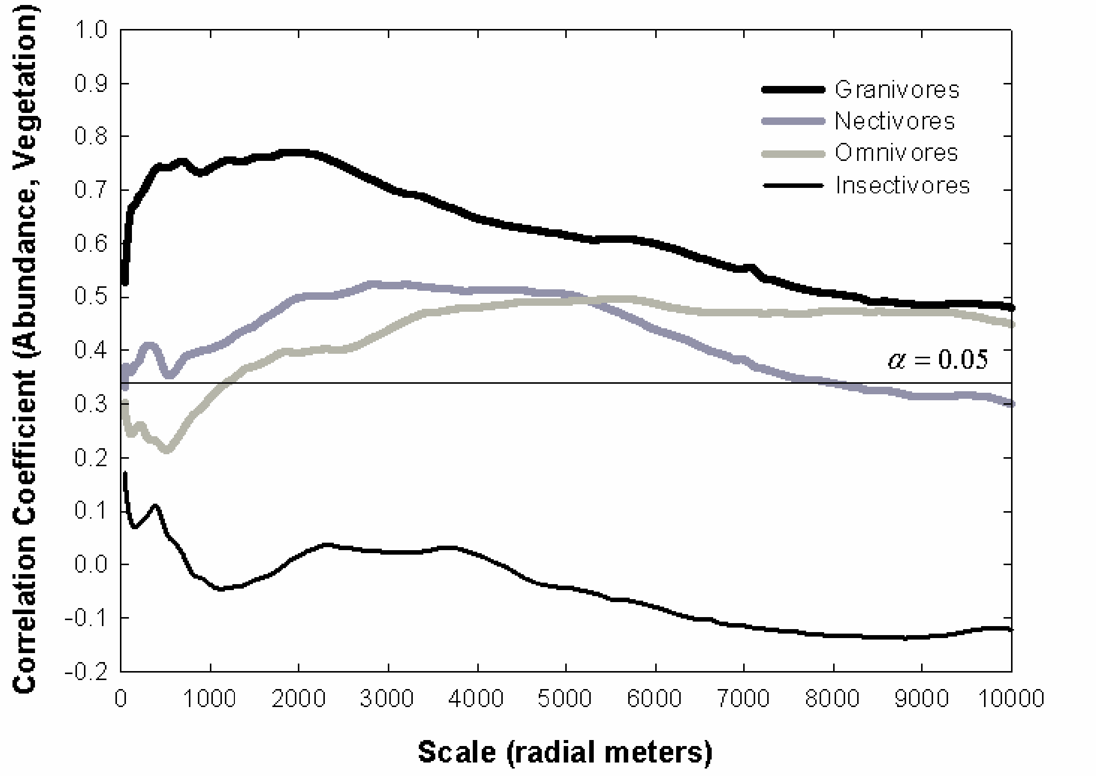

FIG 20. The scalar effects of vegetation on different avian guilds. The y-axis is the correlation coefficient, *r*, between the abundance of individual birds per group and the measured vegetation cover. The x-axis is the radius (m) of the buffer in which the vegetation cover was measured around the census locations. All correlation coefficients are significant above 0.33 at α=0.05.

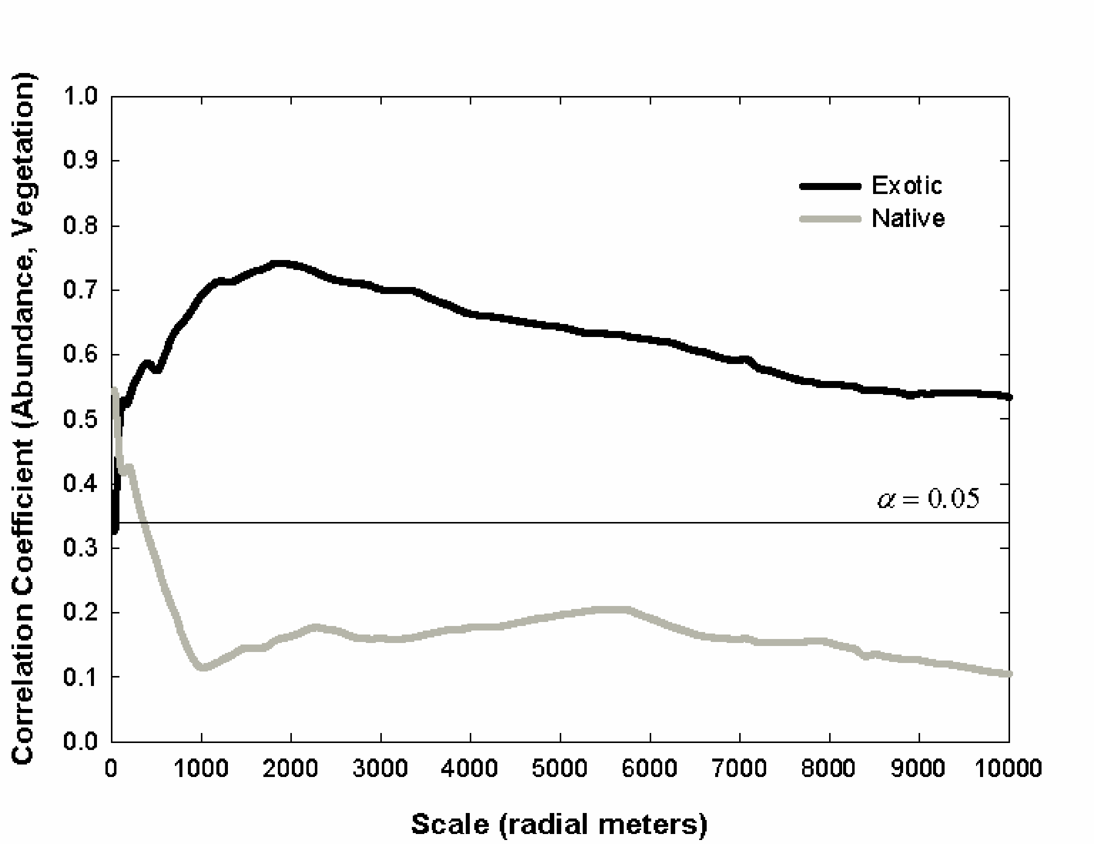

FIG. 21. The scalar effects of vegetation on native vs. exotic species. The y-axis is the correlation coefficient, *r*, between the abundance of individual birds per group and the measured vegetation cover. The x-axis is the radius (m) of the buffer in which the vegetation cover was measured around the census locations. All correlation coefficients are significant above 0.33 at α=0.05.

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