A. PATHWAYS OF THE ECOSYSTEM APPROACH

CHAPTER 4

Quantifying spatiotemporal patterns and ecological effects of urbanization: a multiscale landscape approach

Jianguo Wu[1,2], Alexander Buyantuyev[2], G. Darrel Jenerette[3], Jennifer Litteral[1], Kaesha Neil[1] and Weijun Shen[4]

[1] *School of Life Sciences and Global Institute of Sustainability, Arizona State University, Tempe, Arizona, USA*
[2] *Sino-US Center for Conservation, Energy, and Sustainability Science (SUCCESS), Inner Mongolia University, Hohhot, China*
[3] *Department of Botany and Plant Sciences, University of California, Riverside, California, USA*
[4] *South China Botanical Garden, Chinese Academy of Sciences, Guangzhou, China*

4.1 INTRODUCTION

Urbanization – the spatial expansion of the built environment that is densely packed by people as well as their socioeconomic activities and products – is one of the most prominent features of the modern civilization of humanity. Not so long ago the world was not dominated by *Homo sapiens*, and humans feared and worshiped nature (Chen and Wu 2009). Human domination, however, became the prevailing theme in human society's interactions with nature for more than two centuries particularly after the Industrial Revolution in the 18th century. During this era of frantic acquisition of natural resources, rapid economic growth, and copious technological innovations, the world underwent fundamental sociocultural transformations. As part of the process and outcome, cities are both the symbols of the monumental progress and the evidence of mighty destruction by humanity. In 1800, only 2% of the world's population lived in urban areas, but this number jumped to 14% in 1900 and 30% in 1950. In 2007, the world urban population surpassed the 50% mark, implying that humans have evolved from a predominantly agrarian to a mostly urban species. Although the global population is likely to stabilize around 9.1 billion by 2100, urban populations will continue to increase even after that (Wu 2008a,b).

This increasing urban nature of humanity has resulted in profound environmental implications for the world in the past and the future. Urbanized areas account for about 3% of the Earth's land surface but about 80% of carbon emissions, 60% of residential water use, and close to 80% of the wood used for industrial purposes (Brown 2001, Wu 2008a,b). The ecological

Applied Urban Ecology: A Global Framework, First Edition. Edited by Matthias Richter and Ulrike Weiland.

footprint of a city – measured as the land area necessary for sustaining the current levels of resource consumption and waste discharge by a population – can be hundreds of times as large as its physical size (Rees 1997, Luck and Wu 2002). A number of environmental effects of urbanization have been well documented (Breuste *et al.* 1998, Pickett *et al.* 2001, Wu 2008a). Urbanization influences local climate as impervious surfaces alter surface energy balance to cause temperatures to rise; urbanization leads to excessive consumptions and frequent contamination of water resources; urbanization creates major producers of greenhouse gases and air pollutants that harm both humans and the environment; and urbanization is the most drastic form of land transformation, profoundly influencing biodiversity and ecosystem services.

To deal with these problems, urban ecology is of great necessity and importance. Urban ecological studies date back several decades and the dominant perspectives have evolved in time. Wu (2008a,b) discussed five urban ecological approaches that have stemmed from three broad perspectives on urban ecology: "ecology in cities" (the first approach), "ecology of cities as socioeconomic structures" (the second approach), and "ecology of cities as ecosystems" (the third to fifth approach). The first approach (or the bioecology approach) focuses primarily on the ecology of individual plant and animal species living in urban areas, with little consideration of socioeconomic factors. Some of the earliest urban botanical studies in Europe were prototypical examples (Sukopp 1990). Proposed by social scientists, the second approach (or the socioecology approach) borrows ecological concepts and theories (e.g., niche, competition, succession) to study cities as socioeconomic systems, while biodiversity and ecosystem functioning in the urban system are largely ignored. The perspective of "ecology of cities as ecosystems" recognizes both the socioeconomic and biological components of the urban system, but the degree of integration between the two components varies among the three approaches developed by scientists in different disciplines: the urban systems approach, the urban ecosystem approach, and the urban landscape ecological approach. The urban landscape ecological approach is probably the most promising among all of these because it emphasizes not only the diversity and interactions of the elements of a city but also their spatial patterning and ecological consequences on multiple scales (Pickett *et al.* 1997, Wu 1999, 2008a,b, Grimm *et al.* 2000).

The urban landscape ecological approach is characterized by the explicit emphasis on the relationship between land cover pattern and ecological processes on multiple scales as well as the holistic and humanistic dimensions of the city as a spatially extended system (Wu and David 2002, Wu 2006, 2008a,b, Chen and Wu 2009). The approach includes several interactive components: quantifying the spatiotemporal pattern of urbanization with spatial pattern analysis, analyzing the drivers and mechanisms of urbanization through simulation modeling, and relating urbanization patterns with biodiversity and ecological processes on multiple scales (Figure 4.1). In this chapter we will illustrate some of the elements of the urban landscape ecological approach through a number of studies in the Phoenix metropolitan region, Arizona, the United States (Figure 4.2) – one of the fastest growing metropolizes in this nation and home to the Central Arizona-Phoenix Long-Term Ecological Research project (CAP-LTER) (Grimm *et al.* 2000, Wu *et al.* 2003). The studies discussed here are primarily those carried out by our research group (part of the much larger team of CAP-LTER) since 1997.

The main goal of this chapter is twofold: (1) to illustrate how a landscape ecological approach can be used to study the spatiotemporal pattern and ecological consequences of urbanization, and (2) to provide an overview of the key findings from our urban ecological research in the past decade, ranging from quantifying urban landscape patterns and modeling urban dynamics, to understanding the effects of urbanization on biodiversity and ecosystem processes.

4.2 CHARACTERIZING THE SPATIOTEMPORAL PATTERN OF URBANIZATION

4.2.1 Quantifying urbanization patterns with landscape metrics

Urbanization is fundamentally a spatial process. Socioeconomic decisions drive urban growth, which creates spatial patterns that characterize the different forms of cities. Urban morphology affects, and is affected by, socioeconomic and ecological processes. All natural and human activities vary from location to location across the urban landscape, and this spatial heterogeneity is more salient and profound in cities than any other ecosystems. Thus it is crucial to quantify the spatial and temporal patterns of the urban landscape in order to understand the processes and ecological consequences of urbanization (Pickett *et al.* 1997, Zipperer *et al.* 2000, Luck and Wu 2002, Wu 2008a).

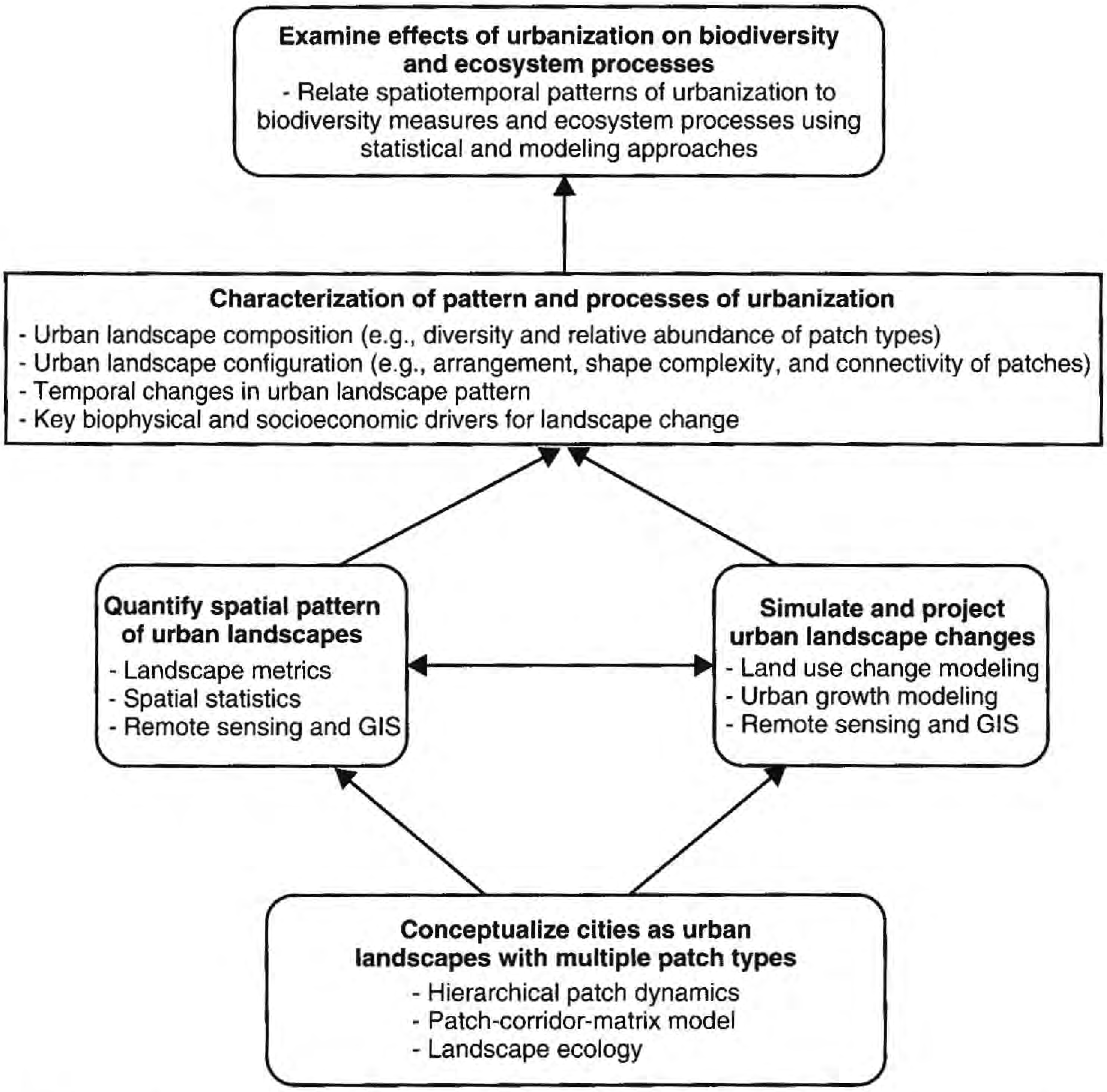

Figure 4.1 Illustration of an urban landscape ecological approach. The urban landscape ecological approach conceptualizes cities as urban landscapes – spatially extended patch mosaics. It emphasizes the relationship between landscape pattern and ecological processes and includes several interactive components: quantifying the spatiotemporal pattern of urbanization with spatial pattern analysis, analyzing the drivers and mechanisms of urbanization through simulation modeling, and relating urbanization patterns with biodiversity and ecological processes.

One of the most commonly used tools in characterizing landscapes of different kinds is known as pattern indices or landscape metrics (McGarigal and Marks 1995, Wu *et al.* 2000, 2002, Wu 2004, Li and Wu 2007). Landscape metrics are synoptic indices that are designed to quantify landscape structural characteristics at three levels: the individual patch (patch-level metrics), the patch type or class (class-level metrics), and the entire landscape encompassing all patches of all types (landscape-level metrics). In general, landscape metrics can be categorized into two groups: composition and configuration metrics. The composition metrics are simple non-spatial measures or summary statistics of the basic features, diversity, and relative abundance of patches of different kinds. Common compositional metrics include the number of patches, patch density (the number of patches per unit area), patch richness (the number of patch types), patch evenness (the degree of uniformity in terms of the relative proportion of each patch type), patch diversity (a combined measure of patch richness and evenness), edge density (the total edge length per unit area), mean patch size, variance in patch size, and the largest patch size. The configuration metrics pertain to the shape, contagion and

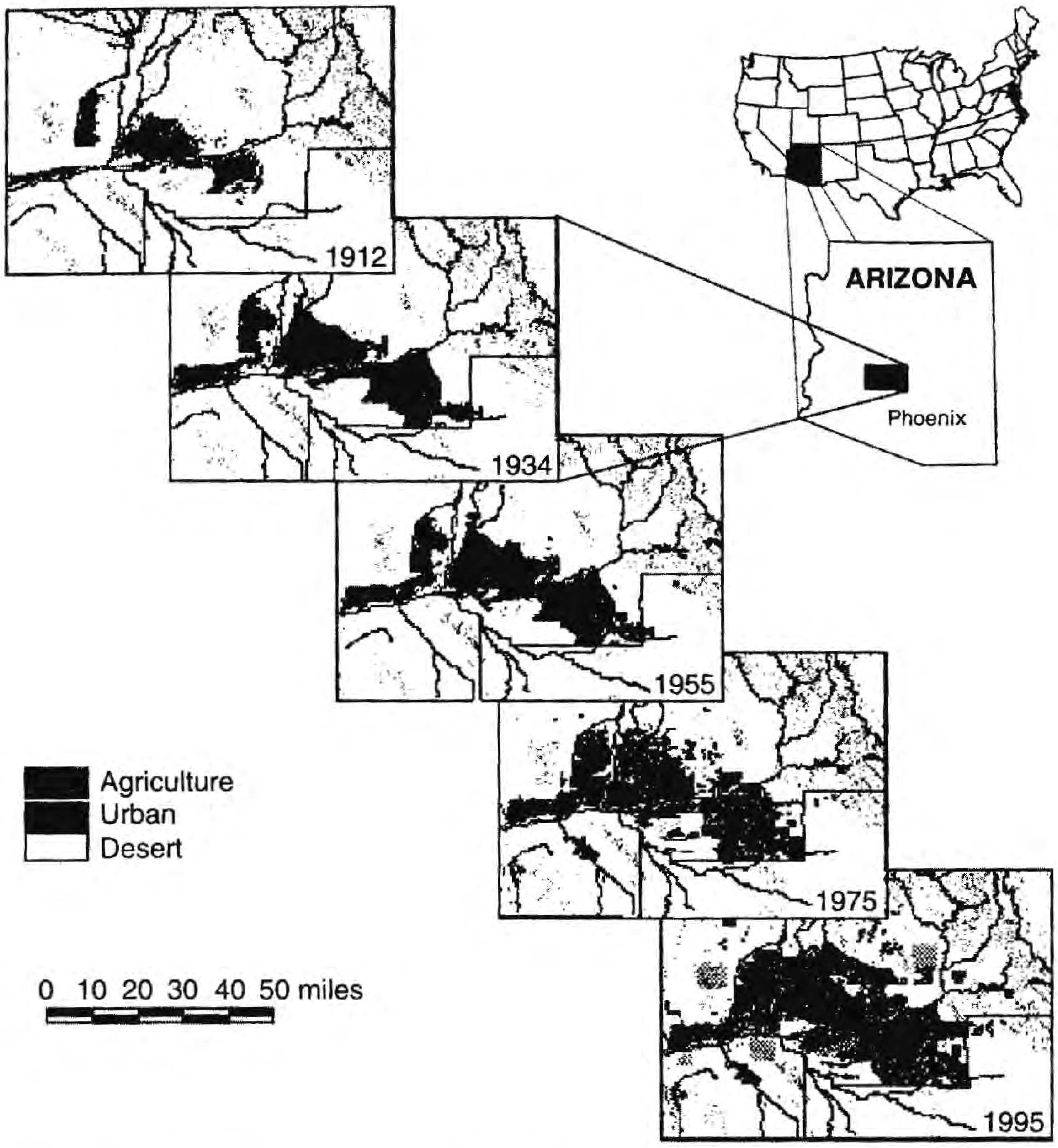

Figure 4.2 Land use change in the Phoenix metropolitan region of Arizona, the United States between 1912 and 1995. The region – home to the Central Arizona-Phoenix Long-Term Ecological Research project (CAP-LTER) – is located in the northern part of the Sonoran desert, where the climate is hot and dry. Average temperature is 30.8 °C during summer and 11.3 °C during winter. Average annual precipitation is 180 mm, with approximately half falling in summer and the other half in winter. The population in this region has increased exponentially from about 20 000 in 1912, 465 000 in 1955, 1.3 million in 1975, 2.4 million in 1995, to over 4 million in 2010. The population growth is highly correlated with the expansion of urbanized area.

interspersion, connectivity and isolation, contrast, and various aspects of spatial arrangement of patches, and thus are usually spatial measures (i.e., indices that contain spatially explicit information). Common configuration metrics include contagion, nearest neighbor, distance, mean patch shape index, landscape shape index, mean patch fractal dimension, landscape fractal dimension, lacunarity, proximity index, patch cohesion index, and interspersion and juxtaposition index (McGarigal and Marks 1995, Wu *et al.* 2000, 2002, Turner *et al.* 2001, Wu 2004).

Quantifying landscape pattern and its change is essential for the monitoring and assessment of ecological consequences of urbanization. Landscape metrics have increasingly been used in the study of urbanization since the 1990s as landscape ecology, urban ecology, geography, and remote sensing frequently come together to deal with common interdisciplinary problems concerning the environment and society. This landscape pattern analysis approach is based on the pattern–process perspective that spatial pattern affects and is affected by underlying processes (Turner *et al.* 1991, Wu and Levin 1994, Wu and Loucks 1995). The application of landscape pattern analysis in urban studies allows for testing hypotheses of how urbanization affects landscape structures and for facilitating the interpretation, assessment and verification of urban models (Wu *et al.* 2000, 2002, 2011, Jenerette and Wu 2001, Herold *et al.* 2003, Wu 2004, Berling-Wolff and Wu 2004a,b, Irwin and Bockstael 2007, Weng 2007).

4.2.2 Other methods for quantifying urban landscape pattern

Urbanization patterns can be quantified with a number of other pre-classification and post-classification change detection methods using multitemporal remote sensing data (Lunetta and Elvidge 1998, Jensen 2004). Pre-classification change detection techniques are based on manipulations with image spectral bands, creation of composites of multidate images that can be classified into change or no-change clusters or analyzed with principal component analysis, simple image band rationing or differencing, image regression, vegetation index differencing, spectral change vector analysis, artificial neural networks, and classification tree analysis (Howarth and Boasson 1983, Ridd and Liu 1998, Rogan *et al.* 2003, Jensen 2004).

Post-classification comparisons based on multidate thematic maps are a common approach in urban ecological studies (Foresman *et al.* 1997, Jensen 2004, Yuan *et al.* 2005). It allows for not only detecting areas of change, but also identifying and quantifying "from–to" transitions that can be converted into probabilities of change used in calibration of land cover change models. While most such studies are pixel-based and prone to significant uncertainties, the object-oriented comparison of maps produced from high-resolution imagery can improve the accuracy of analysis (Ellis *et al.* 2006, Zhou *et al.* 2008).

Spectral mixture analysis (SMA) has been proposed to overcome the problem of mixed pixels, which is common when urban areas are studied using remote sensing data with relatively coarse resolutions (e.g., Landsat imagery). The approach decomposes single pixels linearly into constituent land covers (endmembers) and obtains estimates of their areal fractions at the subpixel level (Small 2005, Buyantuyev and Wu 2007a). The ternary VIS (vegetation–impervious surface–soil) model (Ridd 1995) provides a convenient way of decomposing urban landscapes into a limited number of endmembers. On the other hand, when consideration of the multitude of land covers is desired, one should use the multiple endmember SMA (Rashed *et al.* 2003). Temporal analysis of endmembers has become an efficient approach in quantifying urbanization patterns (Kressler and Steinnocher 1996, Rashed *et al.* 2005, Small 2005).

4.2.3 Effects of scale on the analysis of urban landscape patterns

Urban areas are probably the most spatially heterogeneous among all landscapes, and spatial heterogeneity makes scale a crucial factor in landscape pattern analysis. Most landscape metrics and other spatial analysis methods have been found to be quite sensitive to the scale of analysis (the grain size or spatial resolution and the extent of a map), indicating that landscape pattern is scale dependent (Turner *et al.* 1989, Wickham and Riitters 1995, Jelinski and Wu 1996, Wu *et al.* 2000, 2002, Shen *et al.* 2004, Wu 2004, Shao and Wu 2008). While numerous studies reported on various scale effects in spatial analyses, little was known as to how landscape metrics would change with the scale of analysis prior to 2000. Using data of real and simulated landscapes, our group has systematically explored the scaling relations of landscape metrics since the late 1990s through a series of studies that were focused on the following questions: (i) How does changing grain size or extent affect different landscape metrics for a given landscape? (ii) How does the scaling behavior of various landscape metrics differ among different landscapes? (iii) Are there general scaling relations for certain landscape metrics?

Our results showed that changing grain size and extent had significant effects on both the class- and landscape-level metrics, and these effects fell into two categories (simple scaling functions vs. unpredictable) for class-level metrics, and three categories (simple scaling functions, staircase-like scaling behavior, and unpredictable) for landscape-level metrics (Wu *et al.* 2000, 2002, Shen *et al.* 2004, Wu 2004). Overall, more metrics showed consistent scaling relations with changing grain size than with changing extent at both the class and landscape levels – indicating that effects of changing spatial resolution are generally more predictable than those of changing map sizes. While the same metrics tended to behave similarly at the class level and the landscape level, the scale responses at the class level were much more variable. These results appear robust not only across different landscapes, but also independent of specific map classification schemes (Wu *et al.* 2000, 2002, Shen *et al.* 2004, Wu 2004, Buyantuyev and Wu 2007b, Buyantuyev *et al.* 2010).

These scaling studies have produced new findings that not only help improve our understanding of the scale multiplicity of landscape characteristics, but also have a number of practical implications for dealing with cross-scale problems in heterogeneous landscapes (Wu 2004, Wu *et al.* 2006, Wu 2007). For example, landscape metrics with simple scaling relations reflect those landscape features that can be extrapolated or interpolated across spatial scales readily and accurately using only a few data points. In contrast, unpredictable metrics represent landscape features whose extrapolation is either impossible or requires information on the specifics of the landscape of concern at many different scales. Finally, to quantify urbanization patterns using landscape metrics, it is desirable to use "landscape metric scalograms" – the response curves of landscape metrics to changing grain size or extent, instead of single-scale values (Wu *et al.* 2002, Wu 2004). Such a multiscale approach is crucial for achieving a comprehensive understanding of the spatial complexity of urban landscapes.

4.2.4 Examples from CAP-LTER

We have used landscape pattern analysis extensively in the study of the patterns and processes of urbanization in the Phoenix metropolitan region, Arizona, as part of the CAP-LTER project and other related urban ecological projects. Landscape metrics have been used to quantify the spatial pattern and temporal dynamics of the urban landscape (Wu *et al.* 2000, 2002, 2011, Shen *et al.* 2004, Wu 2004), identify urbanization gradients (Luck and Wu 2002), relate urban landscape features to ecosystem properties (Wu *et al.* 2003, Jenerette *et al.* 2006, 2007, Buyantuyev and Wu 2009, 2010), and to evaluate the projections of urban growth models (Jenerette and Wu 2001, Berling-Wolff and Wu 2004b, Wu *et al.* 2011).

For example, using a combination of landscape metrics and gradient analysis we quantified the center and spatial pattern of the Phoenix metropolitan region (Luck and Wu 2002, Wu *et al.* 2002, Wu 2004). Our research showed that the degree of human modification on the Phoenix urban landscape depended on the distance from the urban center. While the landscape-level metrics were able to identify the center of urbanization, as indicated by the smallest mean patch size and the highest patch richness, density, size coefficient of variation, and landscape shape index, the class-level indices provided more detailed information on the relative contributions of individual land use types (Luck and Wu 2002, Wu 2004).

Different land use types exhibited distinctive but not necessarily unique spatial signatures with different landscape metrics. For instance, for patch type percent coverage, patch density, patch size coefficient of variation, landscape shape index, and area-weighted mean patch shape index, residential and urban land use types displayed similar patterns along the transect from west to the urban center – a largely monotonic gradient with its peak at the urban core (Luck and Wu 2002). Desert showed a similar pattern for patch density, patch size coefficient of variation, landscape shape index, and area-weighted mean patch shape index, but a rather different pattern for patch type percent coverage and mean patch size. For other landscape metrics, agriculture displayed a multiple-peaked pattern. The distinctive "spatial signatures" as distance-based "landscape pattern profiles" may be used to compare urban developmental patterns between cities and dynamics of the same city over time. Such comparisons may help understand different underlying processes that are responsible for various forms of urban morphology (Luck and Wu 2002, Wu *et al.* 2002, 2011, Wu 2004, Seto and Fragkias 2005).

Using historical land use data over a period of about 90 years, Wu *et al.* (2011) used a selected set of landscape metrics to compare urbanization patterns between Phoenix and Las Vegas, the two fastest growing cities in the United States. They found that the two desert cities exhibited markedly similar urbanization patterns: the urban landscape became increasingly more compositionally diverse, structurally fragmented, and geometrically complex as urbanization unfolded. These results can be used to test theories of urban development. For example, Dietzel *et al.* (2005) hypothesized that urbanization exhibits cyclic patterns in time and space driven by two alternating processes: diffusion that spreads urban growth from existing centers to new development areas and coalescence that is characterized by outward expansion and gap infilling of existing urban areas. During a full diffusion–coalescence cycle of urbanization, urban land area increases monotonically; urban patch density, edge density, and mean nearest neighbor distance all increase first, then each peak at different times, and finally decrease, exhibiting a unimodal shape. Contagion is highest at the beginning of the diffusion process and at the end of the coalescence process and reaches its lowest value in between, thus exhibiting a somewhat mirror image of urban patch density (Dietzel *et al.* 2005). Our results, however, showed a monotonic decrease in landscape contagion and a monotonic increase in urban patch density, edge density, and other fragmentation-related metrics over a period of more than 80 years in both Phoenix and Las Vegas. These discrepancies may be attributable to differences in urbanization stages, the scale of analysis, and data accuracy (Wu *et al.* 2011).

Our more detailed studies of urbanization in Phoenix based on Landsat-derived land use/land cover maps between 1985 and 2005 generally supported the general findings of our previous research, showing that urban land covers, especially xeric residential, increased substantially at the expense of undisturbed desert, resulting in a more fragmented and structurally complex landscape (Buyantuyev and Wu 2007a, Buyantuyev *et al.* 2010). In addition, temporal analysis using landscape metrics allowed us to evaluate their sensitivity to changes in desert vegetation and explore various challenges for quantifying urbanization patterns in arid environments (Buyantuyev and Wu 2009, 2010a, Buyantuyev *et al.* 2010).

4.3 SIMULATING SPATIOTEMPORAL DYNAMICS OF URBANIZATION

4.3.1 Importance of simulation models in urban studies

Using landscape metrics or other spatial statistical methods to quantify the spatial and temporal patterns of urbanization is useful in itself, and is often the first step in urban ecological projects (Wu *et al.* 2000, 2003, Luck and Wu 2002). Urbanization, however, is fundamentally driven by socioeconomic processes. Spatial pattern analysis using landscape metrics and other statistical methods does not get to these processes directly although they can be used to help identify the underlying drivers and link patterns and processes in urban landscapes. Simulation modeling of urban spatiotemporal dynamics provides an indispensable tool for exploring the causes and mechanisms of urbanization as well as for managing urban dynamics.

A quantitative model of urbanization provides an efficient method to organize current understanding of urbanization processes into alternate testable hypotheses. Several recent syntheses of urbanization models have documented the progression in both methodology and applications (Jenerette and Wu 2001, Berling-Wolff and Wu 2004a,b, Batty 2005, 2008, Milne *et al.* 2009). This extensive history includes models developed by urban planners, geographers, and landscape ecologists, among others; modeling urban landcover change has a long tradition as an interdisciplinary activity. Early applications of urban growth models were focused on transportation planning (Putnam 1983), followed by applications to better project locations of specific land uses (Harris 1985). Integrations of these two applications were later developed (Berechman and Small 1998), which spawned much of the modern urban spatiotemporal modeling activities (Wu *et al.* 2003, Berling-Wolff and Wu 2004a,b). Current applications of urban growth models include generating improved understanding and projections of urban landcover, potential requirements for water and impacts on water discharges, interactions between urbanization and fire, and effects of urbanization on biodiversity, species invasion, and ecosystem processes (Urban 2000, Jenerette and Wu 2001, Syphard *et al.* 2007, Shen *et al.* 2008, Milne *et al.* 2009).

4.3.2 Approaches to simulating urban dynamics

In developing models of the urbanization process a diverse number of theoretical and computational approaches have been explored. Modeling approaches to urban landscape dynamics range from highly complex descriptions of urban growth with many parameters describing multiple levels of decision making (Landis 1995, Waddell 2002), to cellular automata (CA) models with multiple scales of constraints (Jenerette and Wu 2001, Herold *et al.* 2003, Berling-Wolff and Wu 2004b, Batty 2005), to highly simplified models using a minimal number of parameters (Batty 1991, Makse *et al.* 1995, Fagan *et al.* 2001). Many environmental underlying patterns are important determinants of urbanization trajectories, for example, topography, and socioeconomic patterns, such as transportation corridors and poverty. In addition to these location specific determinants, neighborhood effects of nearby urbanized patches are also important to urban landscape changes. The parameterization schemes vary with theoretical assumptions, and are often based on empirical regressions between predictor variables and landcover patterns or through data-model inversion procedures (Jenerette and Wu 2001, Berling-Wolff and Wu 2004b).

One of the simplest modeling frameworks for urbanization is the spatial Markovian approach that projects land-cover change as a probabilistic outcome (Turner 1987, Wear *et al.* 1996, Jenerette and Wu 2001). Each landscape unit, either a parcel, patch or raster grid cell, is assigned a probability for change based on regression analyses with potential drivers of change. The probabilities can be derived using static variables (e.g., slope) or dynamic variables (e.g., proportion of urbanized area within a defined neighborhood). Another common urban modeling approach is represented by CA models (Batty 1998, Jenerette and Wu 2001, Berling-Wolff and Wu 2004a). CAs are defined by a lattice, a state space, a neighborhood template, and a set of local transition rules. The state space defines the potential states for each cell of the lattice (e.g., urbanized, agricultural, or wildland). The neighborhood template defines the area of influence affecting the transitions in each cell. The local transition rules define the behavior of each cell, which is usually a function of the current state of the cell and the cells in the neighborhood. For urbanization models these rules are often probabilistic.

Hybrid approaches for modeling urban landcover transitions that blend Markov chains with CAs have proliferated in recent decades. In many cases the hybrid approaches allow for explicit inclusion of multiple factors and neighborhoods, which has had much success in generating application-oriented projections and improving understanding of key constraints to urbanization. A number of hybrid models have been developed for specific cities, including UrbanSim for Seattle (Waddell 2002), CUF for San Francisco (Landis 1995), and the SLEUTH model for Santa Barbara with several additional applications globally (Clarke *et al.* 1997, Herold *et al.* 2003). In recent years, agent-based models of urbanization have become increasingly common because of their abilities to simulate directly the decision making processes (Wu and David 2002, Xie *et al.* 2007, Fontaine and Rounsevell 2009, Irwin *et al.* 2009). Each of these hybrid approaches blends multiple theories and techniques for model implementation. They can be tailored to maximize future forecasts, identify consequences of alternative decision scenarios, or improve understanding of the patterns and processes of urbanization.

4.3.3 Examples from CAP-LTER

Several modeling studies of urbanization in the Phoenix metropolitan region have been conducted as part of the CAP-LTER and other related urban projects (Jenerette and Wu 2001, Wu and David 2002, Wu *et al.* 2003, Berling-Wolff and Wu 2004b). The conceptual framework for our modeling work is the hierarchical patch dynamics paradigm (Wu and Loucks 1995, Wu 1999), which is well suited for studying the spatiotemporal dynamics of urbanization and its effects on ecological processes. Our urban development models combined CA-based and spatial Markovan approaches, with extensions to include hierarchical levels of constraints on model dynamics. Defining neighborhood rules for urban models has been challenging, as the rules do not necessarily correspond to easily measured empirical patterns. To ameliorate this problem, Jenerette and Wu (2001) developed a genetic algorithm-based inversion approach that estimates appropriate parameters by iteratively comparing modeled and observed spatial patterns. This technique can be used for parameter estimation and calibration for other land use change models.

Our modeling work has shown that the choice of an appropriate scale (in terms of spatial and temporal resolutions) in urban growth modeling is critically important (Jenerette and Wu 2001, Berling-Wolff and Wu 2004b). If the scale is too fine, data-related uncertainties and overwhelming computational demands can considerably reduce the accuracy and usefulness of the models. At the other extreme, if the scale is too coarse, the model does not have sufficient spatial and mechanistic details that are relevant to socioeconomic drivers and decision making processes. For a given modeling objective, a proper scale (or a range of scales) can be determined by a limited number of simulations on different scales, whose performance is evaluated using landscape metrics and multiple resolution goodness-of-fit (Berling-Wolff and Wu 2004b). Our simulations based on different scenarios suggest that much of the Phoenix area will be urbanized in about 30 years unless dramatic actions are taken soon to slow down the population growth and to build up instead of building out (Berling-Wolff and Wu 2004b). Much of our previous work has emphasized developing useful frameworks for describing the dynamic spatial structure of the Phoenix metropolitan region and developing projections of future patterns. Next generation models of urban change in this region are being developed for extending science applications to better gauge and explore urban sustainability (Wu 2008a,b).

While these examples signify the large strides in urban landcover change modeling, many challenges to describing urban trajectories still exist and may represent barriers for predicting the future of coupled social and biophysical systems (Wu and Marceau 2002, Irwin *et al.* 2009, Milne *et al.* 2009). One major challenge is the inherent multiple-scale nature of land-cover change in urbanizing environments. The decision for converting a given parcel of land to urbanized landcover is directly associated with individuals, developers, municipalities, and banking organizations. For each of these unique agents, the decision making process can vary in response to individual preference to global economic trends. While much urbanization may be contagious at some scales, it may also be dispersive at others. Reconciling the range of scales in the urbanization process and the decision making associated with the process is a fundamental challenge for developing appropriate urban land-cover change models.

Another challenge is the inherent self-organizational, contingent, and adaptive nature of societies and coupled socioecological systems. The rules governing land-cover change may vary systematically and abruptly. The 2008–2009 housing market collapse in the United States epitomizes the occasionally rapid transformation in the process of urban land-cover change. Individual localities may develop rules independent of neighbors or larger systems in which they are embedded. Many of these uncertainties are intractable and can only be poorly predicted at best. Nevertheless, urban land-cover change modeling still has an important role in developing projections of scenarios to improve understanding of how the unpredictable future may unfold. In other words, the inability to predict with confidence does not deplete the importance of developing decision support tools based on reliable science.

4.4 EFFECTS OF URBANIZATION ON BIODIVERSITY AND ECOSYSTEM PROCESSES: EXAMPLES FROM CAP-LTER

4.4.1 Effects of urbanization on biodiversity

Urban habitats are not a random sampling of regions on earth, but tend to be areas of inherently high species richness, making potential losses disproportionately high (Marzluff *et al.* 1998). Urbanization has drastically changed the land use and land cover pattern in the Phoenix metropolitan region, decreasing average patch size and increasing patch density and the juxtaposition of highly contrasting patches (Luck and Wu 2002, Buyantuyev *et al.* 2010, Wu *et al.* 2011). Generally urbanization has the effect of decreasing the amount of habitat for native species and increasing habitat fragmentation for most if not all species. The actual effects of these habitat changes on biodiversity, however, are variable. Humans impose both top-down and bottom-up controls on urban biodiversity. At the CAP-LTER, biodiversity studies have focused mainly on the patterns of three taxonomic groups: perennial plants, arthropods and birds.

Plants

In wildlands, geomorphic and climatological factors generally determine the composition of the vegetative community. In cities, however, humans decide what and where to plant, and seasonal and yearly fluctuations are largely diluted by artificial irrigation practices (Buyantuyev and Wu 2007a, 2009). In studies of perennial plant diversity in the region, alpha diversity was similar among desert and urban sites, but urban sites contained twice as many exotic genera (Hope *et al.* 2006). Beta diversity (turnover) was significantly higher among urban sites leading to an increase in regional gamma diversity (Hope *et al.* 2003, 2006).

Spatial autocorrelation of plant diversity was present among undeveloped Sonoran Desert sites, probably due to factors underlying plant distribution such as slope, aspect, soil type and water availability, but was absent from urban sites (Hope *et al.* 2003, 2006). In urban areas, the best predictors of plant diversity were both ecological and socioecomonic, including family income, median housing age, and land use history (whether it had ever been farmed). Particularly strong was the positive relationship between biodiversity and family income, termed the "luxury effect" (Hope *et al.* 2003).

Arthropods

Arthropod communities have also been found to be affected greatly by changes to their habitats caused by urbanization in the Phoenix metropolitan region. Again, alpha diversity was found to be relatively similar between desert and urban sampling sites, but the species composition was different (McIntyre *et al.* 2001). Ants, springtails, and mites were ubiquitous and abundant across all sites, but the species making up the remainder of each community seemed to be dictated by percent ground cover of a variety of habitat features such as agricultural crops, gravel and lawn, exotic trees and shrubs, native plants and built structures (McIntyre *et al.* 2001).

Related to species composition, the trophic structure of communities in different land uses was also different. Predators, herbivores and detritivores were more common in agricultural sites, whereas omnivores were more abundant in desert, residential and industrial sites (McIntyre *et al.* 2001, Cook and Faeth 2006). The increased productivity in agricultural areas and urban yards clearly had an effect on the temporal patterns observed in arthropod communities in this desert city. Arthropods, being ectotherms, were impacted by seasonal temperature changes, but artificial irrigation in mesic areas seemed to buffer the communities from the effects of seasonal fluctuations in water availability (McIntyre *et al.* 2001).

Birds

Many studies have shown that generally bird diversity is lower but species abundances are higher in urban land uses than in surrounding undeveloped areas (Blair 1996, Cam *et al.* 2000, Johnston 2001, Marzluff and Ewing 2001). In the Phoenix metropolitan region, bird communities were found to be dominated by a few abundant species adept at utilizing urban resources and dealing with urban stressors (Hostetler 1999, Shochat *et al.* 2004, Fokidis *et al.* 2009). Similarly to arthropods, the composition of bird communities was significantly affected by elements of the local habitat structure, particularly volume of woody vegetation (Green and Baker 2003, Hostetler and Knowles-Yanez 2003). While native and non-native woody vegetation affected different guilds differently, native residents and neotropical migrants were negatively affected by urbanization factors such as housing and road density (Green and Baker 2003).

Some of these species have found refuge in desert habitat fragments which consist of more than 20 mountain reserves throughout the Phoenix metropolitan region. However, as in the case of arthropods, these habitat fragments do not completely mimic the outlying desert, and the bird communities seem to be slightly different (Litteral and Wu unpublished data). Even among the fragments themselves, avian community composition differed and appeared to be affected by the size of the fragment, its isolation from other fragments and the type of urban land use surrounding it. The presence of artificial water in the city seemed to shift the balance of competitive relationships towards the proliferation of synanthropic species, such as house sparrows and various species of doves, who are more efficient foragers, but may be less drought tolerant (Litteral and Wu unpublished data).

An integrated approach to studying the changes in the distribution of biodiversity of different taxa has also allowed CAP-LTER researchers to explore trophic interactions that emerge from these changing patterns. For example, the decrease in predation pressure on birds may have been responsible for the increase in the abundance of certain avian species (Faeth *et al.* 2005, Anderies *et al.* 2007). In the same time, the top-down control of arthropod herbivores by avian predators was much stronger in urban than rural areas (Faeth *et al.* 2005, Marussich and Faeth 2009). We also attempted to improve our understanding of how urbanization factors would influence biodiversity patterns by correlating species and community measures with land use and land cover data. We found that the categories used by planners often were not the most relevant categories for biodiversity conservation planning (Hostetler and Knowles-Yanez 2003, Litteral and Wu unpublished data).

4.4.2 Effects of urbanization on soil biogeochemical patterns

Urbanization has a substantial effect on soil biogeochemical patterns and processes (Pouyat *et al.* 2002, Kaye *et al.* 2004, Zhu and Carreiro 2004, Lorenz and Kandeler 2005, Jenerette *et al.* 2006, Hall *et al.* 2009, Pavao-Zuckerman and Byrne 2009). In general, urbanization is associated with increases in soil carbon, organic matter, nitrogen, and other elemental pools. Many human made compounds without native analogs can be found in urban soils ranging from biological control agents, novel pollutants, hormones, to nanoparticles. These materials can be incorporated into soils through irrigation or atmospheric deposition. Gaseous fluxes of carbon and nitrogen compounds from soils are often elevated in urbanized regions. Urban soils are affected by alternate physical substrate, altered climates, altered species, compaction, aeration, novel deposition loads, and other driving factors affecting soil formation (Jenerette *et al.* 2006, Pavao-Zuckerman and Byrne 2009).

While these findings describe a general response pattern of urbanization on soil biogeochemical properties, urbanization also introduces strong heterogeneities in the spatial distribution of urban soils. Strong patterns of urban impacts occur within individual cities and across continental climate gradients of cities. Within an individual city, while mean pools and fluxes increase from background patterns, perhaps more striking are increases in the variability of soil biogeochemical patterns (Jenerette *et al.* 2006). Urban soils within a city range from completely denuded, covered with concrete or buildings, remnant patches of native soils, minimally managed lawnscapes, to highly fertilized, irrigated, and organic matter amended garden soils. This variation can have a distinct spatial structure from the native soil biogeochemical patterns. Within the Phoenix metropolitan region soils were homogenized at small scales and diversified at broader scales when compared to native soils. These contrasting scale-effects may be a signature of urbanization on soil biogeochemical patterns within a city.

Across multiple cities the effects of urbanization on soil biogeochemical patterns may also vary, and describing these broad gradients in urbanization is a growing research direction (Grimm *et al.* 2008a, Pavao-Zuckerman and Byrne 2009). A hypothesis with some support suggests urbanization will cause larger changes to soil biogeochemical patterns in more arid environments than in humid environments (Grimm *et al.* 2008a). However, hypotheses describing urbanization patterns across gradients are only beginning to be developed.

4.4.3 Effects of urbanization on net primary production

Vegetation cover plays a key role in sustaining ecosystem functioning in urban landscapes and providing important ecosystem services (Wu 2008a,b). Net primary production (NPP), the rate at which plant biomass accumulates in an ecosystem, has widely been used as an integrative measure of ecosystem functioning (Lieth and Whittaker 1975, McNaughton *et al.* 1989, Bai *et al.* 2004). While most previous studies reported that urbanization decreased NPP (Milesi *et al.* 2003, Imhoff *et al.* 2004, Xu *et al.* 2007), we found that urbanization actually increased aboveground biomass production in the Phoenix metropolitan region. In general, urbanization enhanced aboveground NPP due to highly productive irrigated plant communities (Buyantuyev and Wu 2009).

The effects of urbanization on vegetation and NPP varied both spatially and temporally. While urban and agricultural land covers exhibited higher aboveground production per unit area in normal and dry years, natural desert and riparian vegetation together contributed more to the regional aboveground NPP in wet years. In particular, during wet years NPP of desert communities dominated by Creosote bush (*Larrea tridentata*) and Bursage (*Ambrosia dumosa*) nearly doubled, whereas urban and agricultural land covers did not respond greatly to rainfall changes. Our correlation analysis confirmed that human supplementation of resources (water and nutrients) in the Phoenix metropolitan region effectively decoupled the usually tight relationship between vegetation growth and precipitation commonly found in arid and semiarid environments (Buyantuyev and Wu 2009). These findings have important implications for predicting long-term environmental impacts in the face of accelerating urbanization and future climate changes.

4.4.4 Effects of urbanization on vegetation phenology

Landscape phenology or vegetation phenology, which studies vegetation development phases and environmental triggers at multiple scales in heterogeneous landscapes, is important for understanding how urbanization affects ecological processes and climate (Liang and Schwartz 2009, Morisette *et al.* 2009). Among common environmental triggers are temperature and moisture, and water availability is particularly important in arid regions (Bowers and Dimmitt 1994, Schwinning *et al.* 2004, Neil and Wu 2006). Excessive heating in urban areas may promote earlier winter-spring growth while simultaneously shortening the overall growing period. Urban land transformations directly affect hydrological flowpaths and consequently vegetation phenology in cities. Also, urban vegetation frequently includes exotic species that may exhibit phenological patterns different from native plants. Studies based on remote-sensing analyses revealed earlier green-up, delayed dormancy, and the extension of growing period by as many as 15 days in some cities of the Northern Hemisphere (Zhang *et al.* 2004a,b).

In the Phoenix metropolitan region our studies have shown that urbanization has affected both leafing and flowering phenology (Buyantuyev and Wu unpublished data, Neil *et al.* 2010, and unpublished data).

Urban vegetation covers (except cultivated grasses) tend to green-up faster than natural desert, and stay photosynthetically active for significantly longer periods. Also, urbanization has added a greater diversity of phenological patterns, some of which are little influenced by climate. Based on herbarium records of plants, we also found that about 19% of plant species examined either advanced or delayed their flowering, and that quite a few species showed significant differences in flowering phenology between urban and non-urban areas (Neil *et al.* 2010). In addition, Neil *et al.* (unpublished data) further showed that landcover types, which were correlated with surface temperatures, had a much stronger effect on plant flowering phenology than water availability in the Phoenix metropolitan region.

4.4.5 Urban heat islands and ecological effects

The urban heat island (UHI) refers to the phenomenon that cities tend to have higher air and surface temperatures than their rural surroundings (Oke 1982, 1997, Arnfield 2003). UHI develops during urbanization as natural vegetation is replaced by impervious surfaces (concrete, asphalt, roof tops, and building walls). This land transformation modifies the near-surface energy budget by reducing evapotranspiration, crowding solar energy absorbing surfaces, and creating heat-trapping canyon-like urban morphology. UHI can be a significant factor for local and regional climatic and environmental changes, leading to a number of social and ecological consequences (Wu 2008a,b, Grimm *et al.* 2008b).

Rapid urbanization in the Phoenix metropolitan region has resulted in increases in the mean daily air temperature by 3.1 °C and in nighttime minimum temperature by 5 °C during the past several decades (Brazel *et al.* 2000, Baker *et al.* 2002). Several studies have shown that surface temperatures are correlated with land use and land cover pattern as well as socioeconomic factors, such as household income (Hsu 1984, Stabler *et al.* 2005, Brazel *et al.* 2007, Jenerette *et al.* 2007). Based on the previous studies, Buyantuyev and Wu (2010) further quantified diurnal and seasonal surface temperature variations at two spatial scales, and explored the biophysical and socioeconomic factors responsible for temperature variations. Our study revealed the existence of the archipelago structure of night-time surface UHI (SUHI). Although it has typically been portrayed at the spatial scale of an urban–rural gradient over which temperature monotonically decreases with distance away from the urban center, UHI is really a multi-scaled phenomenon. Within a city or metropolitan region, many UHIs of different size, shape, intensity, and temporal dynamics may form over sufficiently large impervious surface patches, and similarly many "urban cool islands" (UCIs) may also exist over vegetated patches in the urban landscape. UHIs and UCIs are multiple-scaled, patchy, and dynamic, together forming a hierarchical patch dynamic system (Wu and Loucks 1995, Wu 2008a).

We also were able to identify the daytime heat sinks in the Phoenix region in both the early summer and the late autumn (Buyantuyev and Wu 2010). The formation of the morning heat sinks may be attributed to a variety of factors, including high thermal inertia of built areas and moisture differences between urban and rural areas. In addition, our analysis using geographically weighted regression confirmed the important role of vegetation and pavements in explaining spatiotemporal variations of surface temperatures. The relationship between surface temperature and landscape features were mediated by socioeconomic factors, so that richer neighborhoods tended to be greener and cooler – another example of the "luxury effect." At night, however, the socioeconomic status of neighborhoods was much less important to surface temperatures (Buyantuyev and Wu 2010). In the Phoenix region, urban warming not only has increased energy consumption for cooling and heat stress on biological organisms, but has also resulted in detrimental social consequences such as elevated crime rate (Baker *et al.* 2002). Our ongoing research will address questions of how best to mitigate UHI effects through integrating ecological, economic, and architectural considerations (Wu 2008a,b).

4.4.6 Ecosystem responses to urbanization-induced environmental changes

Urbanization can alter atmospheric chemistry and climate at local and regional scales. Urban areas are characterized by increasing air temperatures, elevated CO_2 concentration, and enhanced nitrogen deposition (Shen *et al.* 2008, Grimm *et al.* 2008b). In the Phoenix metropolitan region, the near surface CO_2 concentration is 470–555 ppm in the city center and 345–370 ppm at the city outskirts (Idso *et al.* 2001, 2002). The nitrogen deposition rate varies from 7 kg N ha^{-1} yr^{-1} in the upwind southwest of Phoenix to 26 kg N ha^{-1} yr^{-1} in the downwind northeast, with the urban core in between (Fenn *et al.* 2003). How do native plants and ecosystems respond to these urbanization-induced environmental changes? Addressing this question is not only crucial to understanding the ecology of urban ecosystems, but also has immediate implications for predicting the ecological consequences of global climate change.

Because of the complexity of multifactor interactions and the lack of field data, we have taken a simulation modeling approach to tackling this question. We adapted a process-based ecosystem model, originally developed by Reynolds and his associates (Reynolds *et al.* 1997, 2006), and parameterized and validated it for the dominant ecosystem of the Sonoran Desert (Shen *et al.* 2005). The model simulates the dynamics of carbon, nitrogen, and water cycling of a desert ecosystem at a daily time step, with explicit consideration of plant functional types of shrub, subshrub, C_3-winter and C_4-summer annual grasses, perennial grasses, and forbs. Through a series of simulation experiments using the model, we have examined how changes in air temperature, CO_2 concentration, and nitrogen deposition may affect ecosystem processes, such as aboveground net primary productivity (ANPP), soil organic matter (SOM), and soil nitrogen content (N_{soil}) along an urban–rural gradient (Shen *et al.* 2008).

Our simulation results showed that these urbanization-induced environmental changes could lead to a 12–120% increase in ANPP and a 69–180% increase in SOM for the native desert remnants in the urban core, with the largest responses occurring in wet years and the smallest in dry years (Shen *et al.* 2008). Conversely, N_{soil} content was higher in the suburban area than in the outside desert and the urban core, a pattern that is consistent with field observations in New York City (Pouyat *et al.* 2002, Gregg *et al.* 2003). We also found that these environmental changes could lead to a shift in species composition of the native ecosystem. Specifically, the urban core environment with higher air temperature, CO_2 concentration, and N deposition generally favored the growth, and thus increased the abundance, of species that were more responsive to CO_2 enrichment and more capable of using winter rains (e.g., C_3 winter annuals). In contrast, a suburban environment with moderate CO_2 concentration and relatively high nitrogen deposition rate promoted the dominance of evergreen and deciduous shrubs that were less responsive to CO_2 enrichment and more drought resistant. Overall, precipitation controlled the magnitude of ecosystem responses to environmental changes in this arid region.

Based on all that we know about the past and present of the world, our planet in future will be warmer in temperature, surrounded by thicker CO_2, and hit by heavier N-containing pollutants. Conceivably, modern cities provide "living laboratories" for studying possible ecological consequences of future climate change. While field manipulative experiments and long-term observations are absolutely necessary, integrating the urban–rural gradient approach with process-based simulation modeling provides a useful way for interfacing urban ecology with global climate change.

4.5 CONCLUDING REMARKS

Although it has a long history, particularly in Europe, urban ecology traditionally has been characterized by approaches that focus on disciplinary inquiries, single-scale investigations, or systems studies without explicit considerations of spatial heterogeneity. In reality, however, cities are the most spatially heterogeneous ecosystems of all, and they are indeed landscapes that are composed of quite conspicuous patches with different sizes, shapes, contents, and dynamics. As we have demonstrated through a series of studies in the Phoenix metropolitan region, therefore, a landscape ecological approach that emphasizes heterogeneity, pattern–process relations, and scale provides an effective way of studying urban systems.

The ultimate goal of urban ecology is to help achieve urban sustainability. This requires that urban ecological studies not only investigate the "ecology" of cities but also the "sustainability" of cities; not only "research" cities in theory but also "shape" them in action. To achieve this goal, we need to integrate natural and social sciences and adopt a transdisciplinary paradigm, and the landscape ecological approach to urban studies is a promising way to move forward (Wu 2006, 2008a,b, 2010). Built on our previous research, our current and future urban ecological studies will further extend this landscape approach, as illustrated in Figure 4.1, to include key elements of urban sustainability in a landscape ecological framework (Figure 4.3). This extended urban landscape

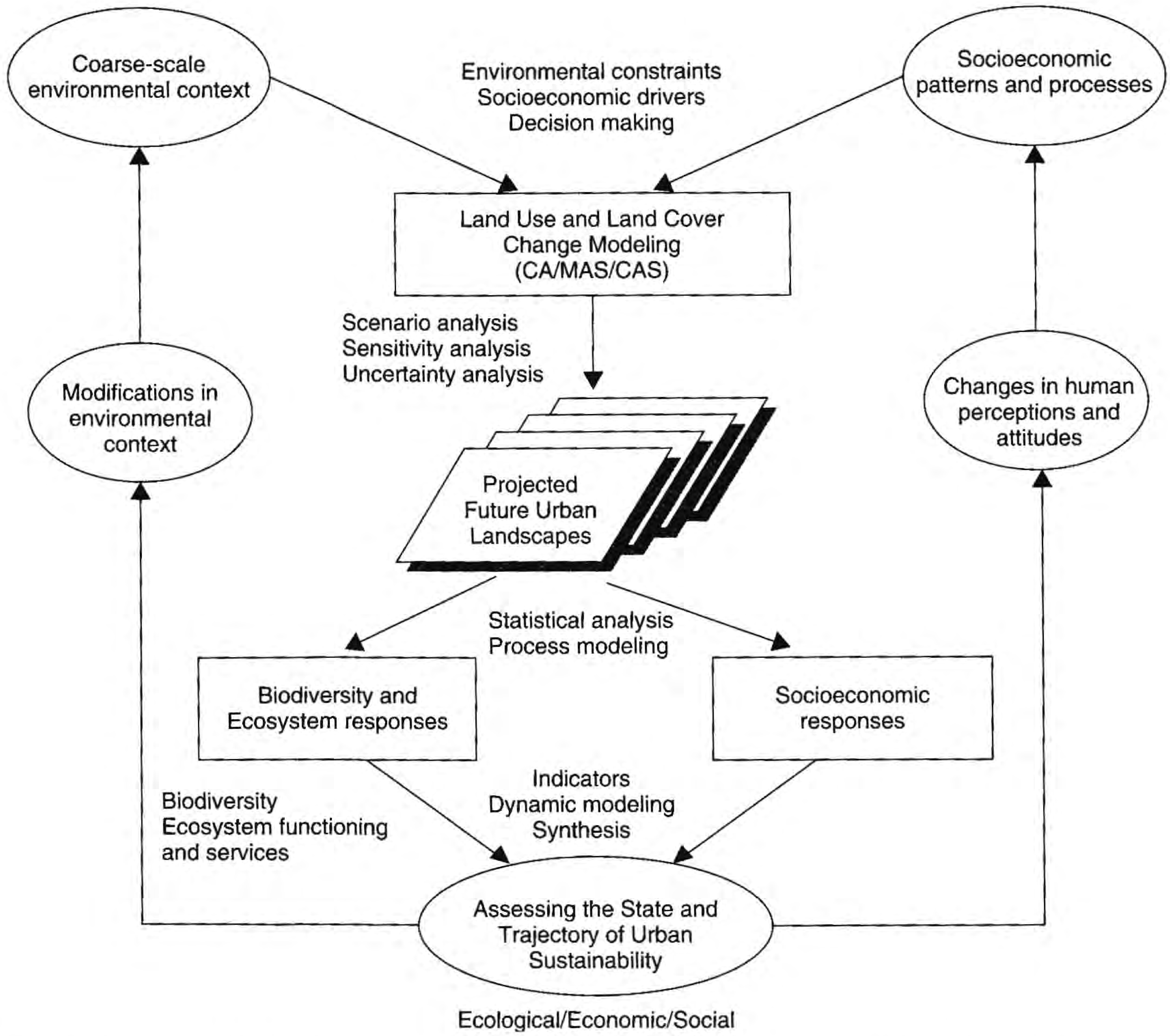

Figure 4.3 An extended urban landscape ecological approach, which integrates ecology with key elements of urban sustainability.

ecological approach will be more effective for coupling natural and social sciences, incorporating feedbacks between urbanization and ecology, and thus providing alternative solutions for decision making.

Cities have been the engines of economic development, cradles of innovation and knowledge production, and centers of sociocultural transformations. Cities also have a lower per capita cost of providing clean water, sanitation, electricity, waste collection, and telecommunications, while offering better access to education, jobs, health care, and social services than rural areas. Cities take up merely 3% of the Earth's land surface, but accommodate more than half the world's population. The potential to increase population density in existing cities without further urban sprawl is great if future urban development focuses on compactness and quality. All of these are important factors for the development of urban sustainability. As the human population continues to rise, the world will be increasingly urban and our well-being and prosperity will increasingly depend on the health of cities. Urban ecology is expected to play an instrumental role in improving existing cities and developing new ones that are more sustainable ecologically, economically, and socially. To realize this goal, urban ecology needs to go beyond the city to consider broader landscapes and go beyond ecology to embrace design sciences (Wu 2008, 2010).

ACKNOWLEDGMENTS

We thank all the current and past members of the Landscape Ecology and Sustainability Laboratory (LESL, formerly LEML) at Arizona State University for their help with and participation in the research reviewed in this chapter. Our research in urban ecology has been supported in part by the National Science Foundation under Grant No. DEB-0423704 (Central Arizona-Phoenix Long-Term Ecological Research) and Grant No. BCS-0508002 (Biocomplexity/CNH), and the US Environmental Protection Agency's Science to Achieve Results (STAR) program (R827676-01-0).

REFERENCES

Anderies, J.M., Katti, M., Shochat, E. (2007) Living in the city: Resource availability, predation, and bird population dynamics in urban areas. *Journal of Theoretical Biology* 247, 36–49.

Annan, K. (2002) United Nations Press Release SG/SM/8261.

Arnfield, A.J. (2003) Two decades of urban climate research: A review of turbulence, exchanges of energy and water, and the urban heat island. *International Journal of Climatology* 23, 1–26.

Bai, Y., Han, X., Wu, J., Chen, Z., Li, L. (2004) Ecosystem stability and compensatory effects in the Inner Mongolia Grassland. *Nature* 431, 181–184.

Baker, L.A., Brazel, A.J., Selover, N., *et al.* (2002) Urbanization and warming of Phoenix (Arizona, USA): Impacts, feedbacks and mitigation. *Urban Ecosystems* 6, 183–203.

Batty, M. (1991) Generating Urban Forms from Diffusive Growth. *Environment and Planning A* 23, 511–544.

Batty, M. (1998) Urban evolution on the desktop: simulation with the use of extended cellular automata. *Environment and Planning A* 30, 1943–1967.

Batty, M. (2005) Agents, cells, and cities: new representational models for simulating multiscale urban dynamics. *Environment and Planning A* 37, 1373–1394.

Batty, M. (2008) The size, scale, and shape of cities. *Science* 319, 769–771.

Berechman, J., Small, K.A. (1998) Research policy and review 25. Modeling land use and transportation: An interpretive review for growth areas. *Environment and Planning A* 20, 1285–1309.

Berling-Wolff, S., Wu, J. (2004a) Modeling urban landscape dynamics: A review. *Ecological Research* 19, 119–129.

Berling-Wolff, S., Wu, J. (2004b) Modeling urban landscape dynamics: A case study in Phoenix, USA. *Urban Ecosystems* 7, 215–240.

Blair, R.B. (1996) Land use and avian species diversity along an urban gradient. *Ecological Applications* 6, 506–519.

Bowers, J.E., Dimmitt, M.A. (1994) Flowering phenology of 6 woody-plants in the northern Sonoran Desert. *Bulletin of the Torrey Botanical Club* 121, 215–229.

Brazel, A.J., Selover, N., Vose, R., Heisler, G. (2000) The tale of two climates – Baltimore and Phoenix urban LTER sites. *Climate Research* 15, 123–135.

Brazel, A., Gober, P., Lee, S.J., *et al.* (2007) Determinants of changes in the regional urban heat island in metropolitan Phoenix (Arizona, USA) between 1990 and 2004. *Climate Research* 33, 171–182.

Breuste, J., Feldmann, H., Uhlmann, O., (eds) (1998) *Urban Ecology*. Springer, Berlin.

Brown, L.R. (2001) *Eco-Economy: Building an Economy for the Earth*. W.W. Norton & Co., New York.

Buyantuyev, A., Wu, J.G. (2007a) Estimating vegetation cover in an urban environment based on Landsat ETM+ imagery: A case study in Phoenix, USA. *International Journal of Remote Sensing* 28, 269–291.

Buyantuyev, A., Wu, J.G. (2007b) Effects of thematic resolution on landscape pattern analysis. *Landscape Ecology* 22, 7–13.

Buyantuyev, A., Wu, J.G. (2009) Urbanization alters spatiotemporal patterns of ecosystem primary productivity: A case study of the Phoenix metropolitan region, USA. *Journal of Arid Environments* 73, 512–520.

Buyantuyev, A., Wu, J.G. (2010) Urban heat islands and landscape heterogeneity: linking spatiotemporal variations in surface temperatures to land-cover and socioeconomic patterns. *Landscape Ecology* 25, 17–33.

Buyantuyev, A., Gries, C., Wu, J. (2010) Multiscale analysis of the urbanization pattern of the Phoenix Metropolitan Landscape of USA: Time, space and thematic resolution. *Landscape and Urban Planning* 94, 206–217.

Cam, E., Nichols, J.D., Sauer, J.R., Hines, J.E., Flather, C.H. (2000) Relative species richness and community completeness: birds and urbanization in the Mid-Atlantic states. *Ecological Applications* 10, 1196–1210.

Chen, X., Wu, J. (2009) Sustainable landscape architecture: implications of the Chinese philosophy of "unity of man with nature" and beyond. *Landscape Ecology* 24, 1015–1026.

Clarke, K.C., Hoppen, S., Gaydos, L. (1997) A self-modifying cellular automaton model of historical urbanization in the San Francisco Bay area. *Environment and Planning B: Planning and Design* 24, 247–261.

Cook, W.M., Faeth, S.H. (2006) Irrigation and land use drive ground arthropod community patterns in an urban desert. *Environmental Entomology* 35, 1532–1540.

Dietzel, C., Herold, M., Hemphill, J.J., Clarke, K.C. (2005) Spatio-temporal dynamics in California's Central Valley: empirical links to urban theory. *International Journal of Geographical Information Science* 19, 175–195.

Ellis, E.C., Wang, H.Q., Xiao, H.S., *et al.* (2006) Measuring long-term ecological changes in densely populated landscapes using current and historical high resolution imagery. *Remote Sensing of Environment* 100, 457–473.

Faeth, S.H., Warren, P.S., Shochat, E., Marussich, W.A. (2005) Trophic dynamics in urban communities. *Bioscience* 55, 399–407.

Fagan, W.F., Meir, E., Carroll, S.S., Wu, J.G. (2001) The ecology of urban landscapes: modeling housing starts as a density-dependent colonization process. *Landscape Ecology* 16, 33–39.

Fenn, M.E., Haeuber, R.A., Tonnesen, G.S., *et al.* (2003) Nitrogen emissions, deposition, and monitoring in the western United States. *Bioscience* 53, 391–403.

Fokidis, H.B., Orchinik, M., Deviche, P. (2009) Corticosterone and corticosteroid binding globulin in birds: Relation to urbanization in a desert city. *General and Comparative Endocrinology* 160, 259–270.

Fontaine, C.M., Rounsevell, M.D.A. (2009) An agent-based approach to model future residential pressure on a regional landscape. *Landscape Ecology* 24, 1237–1254.

Foresman, T.W., Pickett, S.T.A., Zipperer, W.C. (1997) Methods for spatial and temporal land use and land cover assessment for urban ecosystems and application in the greater Baltimore-Chesapeake region. *Urban Ecosystems* 1, 201–216.

Green, D.M., Baker, M.G. (2003) Urbanization impacts on habitat and bird communities in a Sonoran desert ecosystem. *Landscape and Urban Planning* 63, 225–239.

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

Grimm, N., Grove, J.M., Pickett, S.T.A., Redman, C.L. (2000) Integrated approaches to long-term studies of urban ecological systems. *Bioscience* 50, 571–584.

Grimm, N.B., Foster, D., Groffman, P., *et al.* (2008a) The changing landscape: ecosystem responses to urbanization and pollution across climatic and societal gradients. *Frontiers in Ecology and the Environment* 6, 264–272.

Grimm, N.B., Faeth, S.H., Golubiewski, N.E., *et al.* (2008b) Global change and the ecology of cities. *Science* 319, 756–760.

Hall, S.J., Ahmed, B., Ortiz, P., Davies, R., Sponseller, R.A., Grimm, N.B. (2009) Urbanization alters soil microbial functioning in the Sonoran Desert. *Ecosystems* 12, 654–671.

Harris, B. (1985) Urban simulation models in regional science. *Journal of Regional Science* 25, 545–567.

Herold, M., Goldstein, N.C., Clarke, K.C. (2003) The spatiotemporal form of urban growth: measurement, analysis and modeling. *Remote Sensing of Environment* 86, 286–302.

Hope, D., Gries, C., Casagrande, D., Redman, C.L., Grimm, N.B., Martin, C. (2006) Drivers of spatial variation in plant diversity across the central Arizona - Phoenix ecosystem. *Society & Natural Resources* 19, 101–116.

Hope, D., Gries, C., Zhu, W., *et al.* (2003) Socioeconomics drive urban plant diversity. *Proceedings of the National Academy of Sciences (USA)* 100, 8788–8792.

Hostetler, M. (1999) Scale, birds, and human decisions: a potential for integrative research in urban ecosystems. *Landscape and Urban Planning* 45, 15–19.

Hostetler, M., Knowles-Yanez, K. (2003) Land use, scale, and bird distributions in the Phoenix metropolitan area. *Landscape and Urban Planning* 62, 55–68.

Howarth, P.J., Boasson, E. (1983) Landsat digital enhancements for change detection in urban environments. *Remote Sensing of Environment* 13, 149–160.

Hsu, S.I. (1984) Variation of an urban heat-island in Phoenix. *Professional Geographer* 36, 196–200.

Idso, C.D., Idso, S.B., Balling, R.C. (2001) An intensive two-week study of an urban CO_2 dome in Phoenix, Arizona, USA. *Atmospheric Environment* 35, 995–1000.

Idso, S.B., Idso, C.D., Balling, R.C. (2002) Seasonal and diurnal variations of near-surface atmospheric CO_2 concentration within a residential sector of the urban CO_2 dome of Phoenix, AZ, USA. *Atmospheric Environment* 36, 1655–1660.

Imhoff, M.L., Bounoua, L., DeFries, R., *et al.* (2004) The consequences of urban land transformation on net primary productivity in the United States. *Remote Sensing of Environment* 89, 434–443.

Irwin, E.G., Bockstael, N.E. (2007) The evolution of urban sprawl: Evidence of spatial heterogeneity and increasing land fragmentation. *Proceeding of the National Academy of Sciences of the United States of America* 104, 20672–20677.

Irwin, E.G., Jayaprakash, C., Munroe, D.K. (2009) Towards a comprehensive framework for modeling urban spatial dynamics. *Landscape Ecology* 24, 1223–1236.

Jelinski, D.E., Wu, J.G. (1996) The modifiable areal unit problem and implications for landscape ecology. *Landscape Ecology* 11, 129–140.

Jenerette, G.D., Wu, J.G. (2001) Analysis and simulation of land-use change in the central Arizona-Phoenix region, USA. *Landscape Ecology* 16, 611–626.

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

Jenerette, G.D., Harlan, S.L., Brazel, A., Jones, N., Larsen, L., Stefanov, W.L. (2007) Regional relationships between surface temperature, vegetation, and human settlement in a rapidly urbanizing ecosystem. *Landscape Ecology* 22, 353–365.

Jensen, J.R. (2004) *Introductory Digital Image Processing: A Remote Sensing Perspective. Third Edition.* Prentice Hall, Inc., Old Tappan, NJ, USA.

Johnston, R.F. (2001) Synanthropic birds of North America. In: Marzluff, J.M., Bowman, R., Donnelly, R. (eds) *Avian Ecology and Conservation in an Urbanizing World*. Kluwer Academic Publishers, Boston, pp. 49–67.

Kaye, J.P., Burke, I.C., Mosier, A.R., Guerschman, J.P. (2004) Methane and nitrous oxide fluxes from urban soils to the atmosphere. *Ecological Applications* 14, 975–981.

Kressler, F., Steinnocher, K. (1996) Change detection in urban areas using satellite images and spectral mixture analysis. *International Archives of Photogrammetry and Remote Sensing*. Vol. 31, Part B7, Vienna, Austria, pp. 379–383.

Landis, J.D. (1995) Imagining land use futures: Applying the California urban futures model. *Journal of the American Planning Association* 61, 438–457.

Li, H., Wu, J. (2007) Landscape pattern analysis: Key issues and challenges. In: Wu, J., Hobbs, R.J. (eds) *Key Topics in Landscape Ecology*. Cambridge University Press, Cambridge, UK, pp. 39–61.

Liang, L., Schwartz, M. (2009) Landscape phenology: an integrative approach to seasonal vegetation dynamics. *Landscape Ecology* 24, 465–472.

Lieth, H., Whittaker, R. (1975) *Primary Productivity of the Biosphere*. Springer, New York.

Lorenz, K., Kandeler, E. (2005) Biochemical characterization of urban soil profiles from Stuttgart, Germany. *Soil Biology & Biochemistry* 37, 1373–1385.

Luck, M., Wu, J.G. (2002) A gradient analysis of urban landscape pattern: a case study from the Phoenix metropolitan region, Arizona, USA. *Landscape Ecology* 17, 327–339.

Lunetta, R., Elvidge, C. (1998) *Remote Sensing Change Detection: Environmental monitoring methods and applications*. Ann Arbor Press, Chelsea, MI.

Makse, H.A., Havlin, S., Stanley, H.E. (1995) Modelling urban growth patterns. *Nature* 377, 608–612.

Marussich, W., Faeth, S. (2009) Effects of urbanization on trophic dynamics of arthropod communities on a common desert host plant. *Urban Ecosystems* 12, 265–286.

Marzluff, J.M., Ewing, K. (2001) Restoration of fragmented landscapes for the conservation of birds: A general framework and specific recommendations for urbanizing landscapes. *Restoration Ecology* 9, 280–292.

Marzluff, J.M., Gehlbach, F.R., Manuwal, D.A. (1998) Urban environments: influences on avifauna and challenges for the avian conservationist. In: Marzluff, J.M., Sallabanks, R. (eds) *Avian Conservation: Research and Management*. Island Press, Washington, DC., pp. 283–299.

McGarigal, K., Marks, B.J. (1995) *FRAGSTATS: Spatial Pattern Analysis Program for Quantifying Landscape Structure*. Gen. Tech. Rep. PNW-GTR-351. Pacific Northwest Research Station, USDA-Forest Service, Portland.

McIntyre, N.E., Rango, J., Fagan, W.F., Faeth, S.H. (2001) Ground arthropod community structure in a heterogeneous urban environment. *Landscape and Urban Planning* 52, 257–274.

McNaughton, S.J., Oesterheld, M., Frank, D.A., Williams, K.J. (1989) Ecosystem-level patterns of primary productivity and herbivory in terrestrial habitats. *Nature* 341, 142–144.

Milesi, C., Elvidge, C.D., Nemani, R.R., Running, S.W. (2003) Assessing the impact of urban land development on net primary productivity in the southeastern United States. *Remote Sensing of Environment* 86, 401–410.

Milne, E., Aspinall, R.J., Veldkamp, T.A. (2009) Integrated modeling of natural and social systems in land change science. *Landscape Ecology* 24, 1145–1147.

Morisette, J.T., Richardson, A.D., Knapp, A.K., *et al.* (2009) Tracking the rhythm of the seasons in the face of global change: phenological research in the 21st century. *Frontiers in Ecology and the Environment* 7, 253–260.

Neil, K., Wu, J. (2006) Effects of urbanization on plant flowering phenology. *Urban Ecosystems* 9, 243–257.

Neil, K., Landrum, L., Wu, J. (2010) Effects of urbanization on flowering phenology in the metropolitan Phoenix region of USA: Findings from herbarium records. *Journal of Arid Environments*. 74, 440–444.

Oke, T.R. (1982) The energetic basis of the urban heat island. *Quarterly Journal of Royal Meteorological Society* 108, 1–24.

Oke, T.R. (1997) Urban climates and global change. In: Perry, A., Thompson, R. (eds) *Applied Climatology: Principles and Practices*. Routledge, London, pp. 273–287.

Pavao-Zuckerman, M.A., Byrne, L.B. (2009) Scratching the surface and digging deeper: exploring ecological theories in urban soils. *Urban Ecosystems* 12, 9–20.

Pickett, S.T.A., Burch, W.R., Dalton, S.E., Foresman, T.W., Grove, J.M., Rowntree, R. (1997) A conceptual framework for the study of human ecosystems in urban areas. *Urban Ecosystems* 1, 185–199.

Pickett, S.T.A., Cadenasso, M.L., Grove, J.M., *et al.* (2001) Urban ecological systems: Linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. *Annual Review of Ecology and Systematics* 32, 127–157.

Pouyat, R., Groffman, P., Yesilonis, I., Hernandez, L. (2002) Soil carbon pools and fluxes in urban ecosystems. *Environmental Pollution* 116, 107–118.

Putnam, S. (1983) *Integrated Urban Models*. Pion Limited, London, UK.

Rashed, T., Weeks, J.R., Stow, D., Fugate, D. (2005) Measuring temporal compositions of urban morphology through spectral mixture analysis: toward a soft approach to change analysis in crowded cities. *International Journal of Remote Sensing* 26, 699–718.

Rashed, T., Weeks, J.R., Roberts, D., Rogan, J., Powell, R. (2003) Measuring the physical composition of urban morphology using multiple endmember spectral mixture models. *Photogrammetric Engineering and Remote Sensing* 69, 1011–1020.

Rees, W.E. (1997) Urban ecosystems: The human dimension. *Urban Ecosystems* 1, 63–75.

Reynolds, J.F., Virginia, R.A., Schlesinger, W.H. (1997) Defining functional types for models of desertification. In: Smith, T.M., Shugart, H.H., Woodward, F.I. (eds) *Plant Functional Types: Their Relevance to Ecosystem Properties and Glogal Change*. Cambridge University Press, Cambridge, pp. 195–216.

Reynolds, J.F., Kemp, P.R., Ogle, K., Fernández, R.J., Gao, Q., Wu, J. (2006) Modeling the unique attributes of dryland ecosystems. In: Huenneke, L.F., Havstad, K.M., Schlesinger, W.H. (eds) *Structure and Function of a Chihuahuan Desert Ecosystem*. Oxford University Press, Oxford, UK, pp. 321–353.

Ridd, M.K. (1995) Exploring a V-I-S (Vegetation-Impervious Surface-Soil) model for urban ecosystem analysis through remote-sensing – comparative anatomy for cities. *International Journal of Remote Sensing* 16, 2165–2185.

Ridd, M.K., Liu, J.J. (1998) A comparison of four algorithms for change detection in an urban environment. *Remote Sensing of Environment* 63, 95–100.

Rogan, J., Miller, J., Stow, D., Franklin, J., Levien, L., Fischer, C. (2003) Land-cover change monitoring with classification trees using Landsat TM and ancillary data. *Photogrammetric Engineering and Remote Sensing* 69, 793–804.

Schwinning, S., Sala, O.E., Loik, M.E., Ehleringer, J.R. (2004) Thresholds, memory, and seasonality: understanding pulse dynamics in arid/semi-arid ecosystems. *Oecologia* 141, 191–193.

Seto, K.C., Fragkias, M. (2005) Quantifying spatiotemporal patterns of urban land-use change in four cities of China with time series landscape metrics. *Landscape Ecology* 20, 871–888.

Shao, G.F., Wu, J.G. (2008) On the accuracy of landscape pattern analysis using remote sensing data. *Landscape Ecology* 23, 505–511.

Shen, W., Jenerette, G.D., Wu, J., Gardner, R.H. (2004) Evaluating empirical scaling relations of pattern metrics with simulated landscapes. *Ecography* 27, 459–469.

Shen, W., Wu, J., Grimm, N.B., Hope, D. (2008) Effects of urbanization-induced environmental changes on ecosystem functioning in the Phoenix metropolitan region, USA. *Ecosystems* 11, 138–155.

Shen, W., Wu, J., Kemp, P.R., Reynolds, J.D., Grimm, N.B. (2005) Simulating the dynamics of primary productivity of a Sonoran ecosystem: Model parameterization and validation. *Ecological Modelling* 189, 1–24.

Shochat, E., Lerman, S.B., Katti, M., Lewis, D.B. (2004) Linking optimal foraging behavior to bird community structure in an urban-desert landscape: Field experiments with artificial food patches. *American Naturalist* 164, 232–243.

Small, C. (2005) A global analysis of urban reflectance. *International Journal of Remote Sensing* 26, 661–681.

Stabler, L.B., Martin, C.A., Brazel, A.J. (2005) Microclimates in a desert city were related to land use and vegetation index. *Urban Forestry & Urban Greening* 3, 137–147.

Sukopp, H. (1990) Urban ecology and its application in Europe. In: Sukopp, H., Hejny, S., Kowarik, I. (eds) *Urban Ecology: Plants and Plant Communities in Urban Environments*. SPB Academic Publishing bv, The Hague, The Netherlands, pp. 2–22.

Syphard, A.D., Clarke, K.C., Franklin, J. (2007) Simulating fire frequency and urban growth in southern California coastal shrublands, USA. *Landscape Ecology* 22, 431–445.

Turner, M.G. (1987) Spatial simulation of landscape changes in Georgia: A comparison of 3 transition models. *Landscape Ecology* 1, 29–36.

Turner, M.G., O'Neill, R.V., Gardner, R.H., Milne, B.T. (1989) Effects of changing spatial scale on the analysis of landscape pattern. *Landscape Ecology* 3, 153–162.

Turner, S.J., O'Neill, R.V., Conley, W., Conley, M.R., Humphries, H.C. (1991) Pattern and scale: Statistics for landscape ecology. In: Turner, M.G., Gardner, R.H. (eds) *Quantitative Methods in Landscape Ecology*. Springer, New York, pp. 17–49.

Turner, M.G., Gardner, R.H., O'Neill, R.V. (2001) *Landscape Ecology in Theory and Practice: Pattern and Process*. Springer, New York.

Urban, D.L. (2000) Using model analysis to design monitoring programs for landscape management and impact assessment. *Ecological Applications* 10, 1820–1832.

Waddell, P. (2002) Modeling urban development for land use, transportation, and environmental planning. *Journal of the American Planning Association* 68, 297–314.

Wear, D.N., Turner, M.G., Flamm, R.O. (1996) Ecosystem management with multiple owners: Landscape dynamics in a southern Appalachian watershed. *Ecological Applications* 6, 1173–1188.

Weng, Y.-C. (2007) Spatiotemporal changes of landscape pattern in response to urbanization. *Landscape and Urban Planning* 81, 341–353.

Wickham, J.D., Riitters, K.H. (1995) Sensitivity of landscape metrics to pixel size. *International Journal of Remote Sensing* 16, 3585–3595.

Wu, J. (1999) Hierarchy and scaling: Extrapolating information along a scaling ladder. *Canadian Journal of Remote Sensing* 25, 367–380.

Wu, J. (2004) Effects of changing scale on landscape pattern analysis: Scaling relations. *Landscape Ecology* 19, 125–138.

Wu, J. (2006) Landscape ecology, cross-disciplinarity, and sustainability science. *Landscape Ecology* 21, 1–4.

Wu, J. (2007) Scale and scaling: A cross-disciplinary perspective. In: Wu, J., Hobbs, R. (eds) *Key Topics in Landscape Ecology*. Cambridge University Press, Cambridge, UK, pp. 115–142.

Wu, J. (2008a) Toward a landscape ecology of cities: beyond buildings, trees, and urban forests. In: Carreiro, M.M., Song, Y.C., Wu, J.G. (eds) *Ecology, Planning and Management of Urban Forests: International Perspectives*. Springer, New York, pp. 10–28.

Wu, J. (2008b) Making the case for landscape ecology: an effective approach to urban sustainability. *Landscape Journal* 27, 41–50.

Wu, J. (2010) Urban sustainability: an inevitable goal of landscape research. *Landcape Ecology* 25, 1–4.

Wu, J., David, J.L. (2002) A spatially explicit hierarchical approach to modeling complex ecological systems: Theory and applications. *Ecological Modelling* 153, 7–26.

Wu, J., Levin, S.A. (1994) A spatial patch dynamic modeling approach to pattern and process in an annual grassland. *Ecological Monographs* 64(4), 447–464.

Wu, J., Loucks, O.L. (1995) From balance-of-nature to hierarchical patch dynamics: A paradigm shift in ecology. *Quarterly Review of Biology* 70, 439–466.

Wu, J., Marceau, D.J. (2002) Modeling complex ecological systems: An introduction. *Ecological Modelling* 153, 1–6.

Wu, J., Jenerette, G.D., David, J.L. (2003) Linking land use change with ecosystem processes: A hierarchical patch dynamics model. In: Guhathakurta, S. (ed.) *Integrated Land Use and Environmental Models*. Springer, Berlin, pp. 99–119.

Wu, J., Jelinski, D.E., Luck, M., Tueller, P.T. (2000) Multiscale analysis of landscape heterogeneity: Scale variance and pattern metrics. *Geographic Information Sciences* 6, 6–19.

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

Wu, J., Jones, K.B., Li, H., Loucks, O.L. (eds) (2006) *Scaling and Uncertainty Analysis in Ecology: Methods and Applications*. Springer, Dordrecht, The Netherlands.

Wu, J., Shen, W.J., Sun, W.Z., Tueller, P.T. (2002) Empirical patterns of the effects of changing scale on landscape metrics. *Landscape Ecology* 17, 761–782.

Xie, Y., Batty, M., Zhao, K. (2007) Simulating emergent urban form using agent-based modeling: Desakota in the Suzhou-Wuxian region in China. *Annals of the Association of American Geographers* 97, 477–495.

Xu, C., Liu, M., An, S., Chen, J.M., Yan, P. (2007) Assessing the impact of urbanization on regional net primary productivity in Jiangyin County, China. *Journal of Environmental Management* 85, 597–606.

Yuan, F., Sawaya, K.E., Loeffelholz, B.C., Bauer, M.E. (2005) Land cover classification and change analysis of the Twin Cities (Minnesota) Metropolitan Area by multitemporal Landsat remote sensing. *Remote Sensing of Environment* 98, 317–328.

Zhang, X.Y., Friedl, M.A., Schaaf, C.B., Strahler, A.H. (2004a) Climate controls on vegetation phenological patterns in northern mid- and high latitudes inferred from MODIS data. *Global Change Biology* 10, 1133–1145.

Zhang, X.Y., Friedl, M.A., Schaaf, C.B., Strahler, A.H., Schneider, A. (2004b) The footprint of urban climates on vegetation phenology. *Geophysical Research Letters* 31, L12209, doi:10.1029/2004GL020137.

Zhou, W., Troy, A., Grove, M. (2008) Object-based land cover classification and change analysis in the Baltimore metropolitan area using multitemporal high resolution remote sensing data. *Sensors* 8, 1613–1636.

Zhu, W.X., Carreiro, M.M. (2004) Variations of soluble organic nitrogen and microbial nitrogen in deciduous forest soils along an urban - rural gradient. *Soil Biology & Biochemistry* 36, 279–288.

Zipperer, W.C., Wu, J., Pouyat, R.V., Pickett, S.T.A. (2000) The application of ecological principles to urban and urbanizing landscapes. *Ecological Applications* 10, 685–688.