Journal of Environmental Management 92 (2011) 331–362

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Review

Urban ecological systems: Scientific foundations and a decade of progress

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ARTICLE INFO

Article history:
Received 22 July 2009
Received in revised form
22 July 2010
Accepted 22 August 2010
Available online 20 October 2010

Keywords:
Animal
Biodiversity
City
Design
Ecosystem
Environment
Human
Land use
Plant
Spatial heterogeneity
Socioecological
Suburban
Theory
Urban
Urbanization

ABSTRACT

Urban ecological studies, including focus on cities, suburbs, and exurbs, while having deep roots in the early to mid 20th century, have burgeoned in the last several decades. We use the state factor approach to highlight the role of important aspects of climate, substrate, organisms, relief, and time in differentiating urban from non-urban areas, and for determining heterogeneity within spatially extensive metropolitan areas. In addition to reviewing key findings relevant to each state factor, we note the emergence of tentative "urban syndromes" concerning soils, streams, wildlife and plants, and homogenization of certain ecosystem functions, such as soil organic carbon dynamics. We note the utility of the ecosystem approach, the human ecosystem framework, and watersheds as integrative tools to tie information about multiple state factors together. The organismal component of urban complexes includes the social organization of the human population, and we review key modes by which human populations within urban areas are differentiated, and how such differentiation affects environmentally relevant actions. Emerging syntheses in land change science and ecological urban design are also summarized. The multifaceted frameworks and the growing urban knowledge base do however identify some pressing research needs.

1. The urban world

Urbanization is a dominant demographic trend and an important component of global land transformation. More than half of the planet's population now lives in cities, up 30% from 50 years ago, and urban areas are gaining 67 million people per year. By 2030, approximately 5 billion people are expected to live in urban areas, or 60% of the projected global population of

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doi:10.1016/j.jenvman.2010.08.022

8.3 billion. Over the next 25 years, rural populations are expected to decline, meaning that *all* population growth will occur in urban areas (United Nations Population Fund, 2007). The developed nations have more urbanized populations; for example, close to 80% of the United States' (US) population is urban. Urbanization has also resulted in a dramatic rise in the size of cities: over 300 cities have more than 1,000,000 inhabitants and 20 "megacities" exceed 10 million. The increasing population and spatial prominence of urban areas is reason enough to study them. An even more compelling argument for understanding how cities work ecologically is the need for information for decision makers involved in regional planning or conservation, for example. Proper management of cities will ensure that they are ecologically, economically, and socially more sustainable places to live in the future (Platt, 1994).

In addition to its global reach, urbanization has important effects in regions (Forman, 2008; Grimm et al., 2008). For example, in industrialized nations, the conversion of land from wild and agricultural uses to urban and suburban occupancy is growing at a faster rate than the population in urban complexes. Thus, urban areas increasingly interdigitate with wild lands. Cities are no longer compact, isodiametric aggregations, but rather sprawl in fractal or web-like configurations (Makse et al., 1995; Batty, 2008). Indeed, even for many rapidly growing metropolitan areas, the suburban zones, or "boomburgs" (Lang and LeFurgy, 2007), are growing much faster than other zones (Katz and Bradley, 1999). These new forms of urban development (Fig. 1), including exurbs, edge cities (Garreau, 1991), and housing interspersed in forest, shrubland and desert, bring people possessing urban financial equity, habits, and expectations into daily contact with habitats formerly controlled by

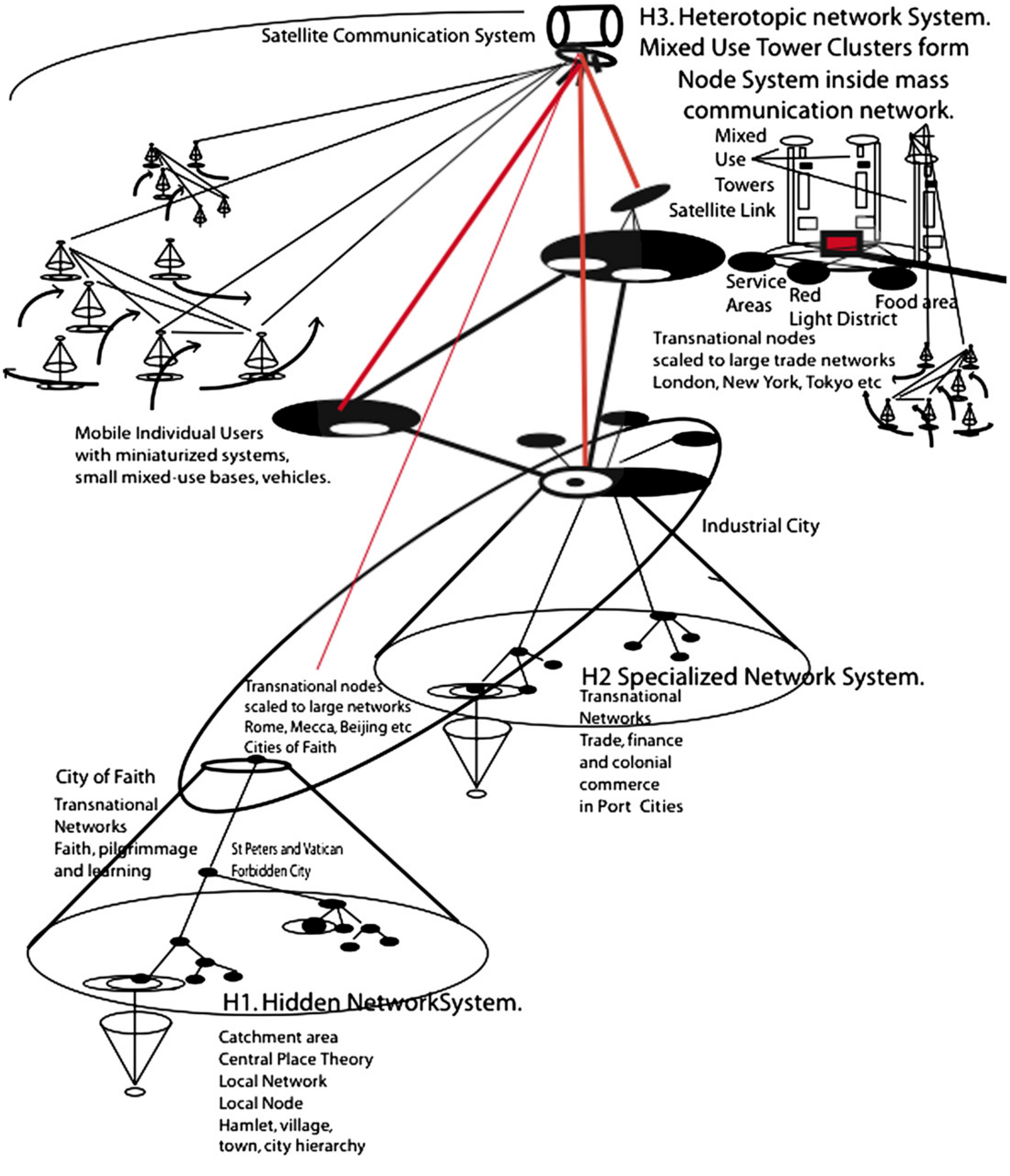

Fig. 1. A new city form representing global connections and communications, and local discontinuities. Four overlapping models operate in different combinations to form the emerging, highly differentiated patchworks, linked together to create a new global city. Compressed layers of networks flattened in the meshwork city can all be present in a particular city simultaneously, represented as a series of patches containing different systems of organization, depending on the power of particular urban actors in each city network or mesh. Used with permission of Prof. G. David Shane.

agriculturalists, foresters and conservationists (Bradley, 1995; Bunce, 1998).

Beyond the practical reasons for increasing ecological knowledge of urban systems, there are benefits to the basic ecological science. Urban systems can serve as model systems for examining the interaction of social and biophysical patterns and processes (Collins et al., 2000; Redman et al., 2004). In addition, many of the changes in urban areas anticipate the otherwise unprecedented alterations that will follow global environmental change in other ecosystems (Grimm et al., 2008).

Although the ecology of urban areas has elicited the scholarly attention of ecologists, physical and social scientists, as well as planners for a long time (e.g., Stearns, 1971; Sukopp et al., 1990), there is much opportunity to extend and integrate knowledge of urban complexes using an ecological lens (Steiner, 2002). The purpose of this paper is to review the status of ecological knowledge of key components of urban areas, and to point toward frameworks for increased integration of ecological, social, and economic research (McDonnell and Pickett, 1993; Grimm et al., 2008). This paper updates and considerably extends a review by several of us nearly a decade ago (Pickett et al., 2001).

1.1. Conceptions of the urban

Urban ecosystems are those in which people live at high densities, and where built structures and infrastructure cover much of the land surface. The US Bureau of the Census defines urban areas as those with a minimum population of 2500 people where densities reach or exceed 1000 people per square mile (2.59 km^2) in the core and at least 500 people per square mile in surrounding Census block groups. However, an ecological understanding of urban systems also must include less densely populated areas in order to capture the full range of urban effects and the existence of thresholds of change across space. Therefore, in the broadest sense, urban ecosystems comprise not only city cores, but suburban areas, sparsely settled villages connected by commuting or utilities to the more densely settled and thoroughly built up areas. Strictly, urban systems do not encompass the hinterlands directly managed or affected by energy and material transformations generated in urban core and suburban lands (Wittig, 2009).

The boundaries of urban ecosystems are often set by watersheds, airsheds, commuting radii, or by administrative units. In other words, boundaries of urban ecosystems are set in the same ways and for the same reasons as are the boundaries of any other ecosystem. In the case of urban ecosystems, it is clear that many fluxes and interactions extend well beyond their boundaries defined for political, research, or biophysical reasons. Urban ecology, as an integrative science with roots in ecology, focuses on urban ecosystems as broadly conceived above. There is little to be gained from seeking distinctions between "urban" and abutting "wild" lands, as a comprehensive, spatially extensive, systems approach is most valuable for science (Pickett et al., 1997a) and management (Rowntree, 1995).

There are two meanings of "urban ecology" in the literature (Sukopp, 1998). One is a scientific definition, and the other emerges from urban planning. In science, the term "urban ecology" has been traditionally used for studies of the distribution and abundance of organisms in and around cities, and on the biogeochemical budgets of urban areas. In planning, urban ecology has focused on designing the environmental amenities of cities for people, and on reducing environmental impacts of urban regions (Deelstra, 1998). The planning perspective is normative and claims ecological justification for specific planning approaches and goals. Even though our focus is primarily on ecological science, urban planning and design provide important context and motivation for ecological research. Therefore, we begin by reviewing key aspects of these complementary approaches.

Ecologists have been surprised by the presence and vitality of organisms in and near cities. In addition, many patterns of distribution and abundance of species in and among cities are regular and repeatable. The patterns and environmental correlates discovered by ecologists, botanists, zoologists, and wildlife biologists are an example of the understanding of ecology *in* cities (Grimm et al., 2000). Associated with such biotic patterns are data on abiotic environmental conditions that affect various parts of the metropolitan matrix. The past decade has seen the maturation of a new approach to urban ecology. Whereas most earlier urban ecological research was focused on the obvious green components in cities, a more comprehensive approach that investigates entire urban complexes as heterogeneous ecological systems – an ecology *of* cities – has solidified (Pickett et al., 1997a; Grimm et al., 2000).

It has taken a long time for a broadly based ecological approach to urban complexes to emerge. Although the first volume of the journal *Ecology* contained a scientific paper devoted to the effect of weather on the spread of pneumonia in the human populations of New York and Boston (Huntington, 1920), the interactions of humans with the urban environment has been primarily the province of planners and landscape architects. For example, Central Park in New York City and other urban parks designed by Frederick Law Olmsted seem intuitively to link environmental properties to human wellbeing in cities. In particular, Olmsted's design for the Boston Fens and Riverway exhibits ecological prescience in its sophisticated combination of wastewater management and recreational amenity (Spirn, 1998). McHarg's (1969) *Design with Nature* alerted planners and architects to the value of incorporating knowledge of ecological and natural features among the usual engineering, economic, and social criteria when developing a regional plan. In McHarg's approach, environmental risks and amenities of different types are mapped on separate layers. Examining the composite map suggests where certain types of development should or should not occur. A still more explicitly ecological approach is that of Spirn (1998), which examines how natural processes are embedded in cities, and how the interaction of the built environment and natural processes affect economy, health, and human community. For instance, she showed how the forgotten environmental template of drainage networks continued to affect both the built and the social structures of a Philadelphia neighborhood.

The ecological planning perspective, often labeled human ecology, is especially strong in Europe (Sukopp, 1998). Planning in Germany has been heavily influenced by a national program of biotope mapping that includes cities (Sukopp, 1990; Werner, 1999). This program includes description of the flora and fauna as a key to identifying habitats that are significant for protecting natural resources, for people's quality of life, and in forming a sense of place and identity in the city (Werner, 1999). In addition to identifying specific biotopes, researchers in Mainz have mapped the distribution of flora and fauna, natural phenomena, and recreational activities within the biotopes (Frey, 1998). Similar research by the Polish Academy of Sciences has focused on urban and suburban areas. The research included studies of soils and abiotic ecosystem components, and research by social scientists in a mosaic of habitats with different degrees of development (Zimny, 1990).

Urban ecology as planning is contrasted with "spatial planning" (de Boer and Dijst, 1998) in which primary motivations are the degree of segregation or aggregation of different economic and social functions, efficiency of transportation and delivery of utilities, and efficient filling of undeveloped space. Additional components of urban planning having ecological foundations include life cycle analysis of products (McDonough and Braungart, 2002), utility planning based on use rather than medium, efficiency of resource use, exploitation of green infrastructure, and requirements for monitoring of the results (Breuste et al., 1998; Benedict and McMahon, 2002).

While such planning research is ecologically motivated, and it maps environmental amenities, it rarely incorporates data on ecological function. It, therefore, relies on general ecological principles and assumptions, and on the success of prior case histories (Flores et al., 1997). The insights of urban ecology as planning are summarized in manuals and codified in zoning and planning practice (Sukopp, 1998). However, like other environmental practices, it may not be securely applicable in novel ecological circumstances. Given the changing forms of cities worldwide (Shane, 2005; Boone and Modarres, 2006) and the reality of climate change, such novel ecological circumstances are in the offing.

2. Features of urban ecosystems

All ecosystems are affected by the same broad suite of state factors (Chapin et al., 2002): 1) the prevailing climate, 2) the substrate, 3) the resident organisms and their residual effects, 4) relief, including elevation, slope, and aspect, and 5) the time over which the first four factors have been acting, which can be summarized as the history of the system. Of course, in urban ecosystems, organisms must include humans and their social and economic manifestations as well as native and introduced plants, animals, and microbes. We focus on urban climates, urban hydrology, urban soils, urban biota, the biogeochemistry of urban systems, and the role of human actors. Urban systems can be useful end members for comparing the effects of altered climates, organismal components, substrates, and land forms (Zipperer et al., 1997; Carreiro et al., 2009). Throughout the review, examples of interaction between human actors and the environmental structures and processes will be given. Following the survey of factors and interactions, we will point out existing and emerging frameworks for integrating this array of interacting factors with each other, and indicate how the changing urban context – an effect of the time factor – is expressed in issues of urban design and planning, urban sustainability, and system resilience. We will include reference to the topographic factor, that is, slope, aspect, and land form, within discussion of other state factors.

3. Urban climates

Urban climate comprises temperature of air and soil, and the hydrological conditions in and around cities. We describe the factors and their causes, and illustrate key effects. Additional effects will emerge in later discussions of biogeochemical processes and human agency.

3.1. The urban heat island: patterns and causes

Heat islands represent the difference between urban and rural temperatures that are directly related to urban land cover and human energy use (Oke, 1995). Cities in mid-latitudes of the US are typically 1–2 °C warmer than the surroundings in winter, and 0.5–1.0 °C warmer in summer (Botkin and Beveridge, 1997). New York City is, on average, 2–3 °C warmer than any other location on a 130 km urban–rural transect (McDonnell et al., 1993). Ambient air temperatures may reach maxima of 5–10 °C warmer than hinterlands (Zipperer et al., 1997). Differences often are negligible in the daytime but develop rapidly after sunset, peaking 2–3 h later when radiative cooling differences between urban and non-urban land covers are maximized (Landsberg, 1981; Voogt and Oke, 2003). These differences are caused by the lower vegetation cover and darker surface materials in the urban landscape (Akbari et al., 2001), which change albedo, thermal capacity and heat conductivity, resulting in a thermal climate that is warmer than surrounding non-urban areas (Voogt and Oke, 2003).

The duration and magnitude of the temperature differential between urban and surrounding non-urban areas depend on the spatial heterogeneity of the urban landscape (Arnfield, 2003). Heterogeneity exists in vegetation, building and paving materials, and the morphology of urban landscapes (e.g., Cadenasso et al., 2007a), which can lead to changes in air temperatures (Akbari et al., 2001; Voogt and Oke, 2003). With the development of thermal remote sensing technology, a variety of remotely-sensed data has been used in studies of urban heat islands (UHI), including NOAA AVHRR (Balling and Brazell, 1988), MODIS (Pu et al., 2006), Landsat TM/ETM+ (Weng et al., 2004), ASTER (Pu et al., 2006) and airborne ATLAS data (Lo et al., 1997). These studies reveal the spatial patterns of UHIs and also investigate the relationship between UHIs and surface cover characteristics such as vegetation abundance measured by NDVI (Normal Difference Vegetation Index) (e.g., Weng et al., 2004) and land cover (e.g., Weng, 2003). However, thermal remote sensing of land surface temperatures and UHIs has largely been limited to qualitative descriptions of thermal patterns and simple correlations with land covers. This is partly due to the tendency to use thematic land-use and land-cover data, rather than quantitative surface descriptors, to represent urban landscapes (Voogt and Oke, 2003). As the percentage of human-made surfaces increases, the temperature differential increases. Hence, the urban core is warmer than neighboring residential areas, which are warmer than neighboring farmlands or forests. Increased temperatures in and around cities enhance ozone formation, the number of officially recognized pollution days, and trace gas emission (Sukopp, 1998).

The heat island varies by region as illustrated by a comparison between Baltimore, Maryland and Phoenix, Arizona (Brazel et al., 2000). During the summer, mean maximum temperatures in Baltimore were greater than the rural landscape. Phoenix, on the other hand, became a summer oasis, with cooler temperatures than the surrounding desert. The cooling of Phoenix is due to the watering of mesic plantings in the city (Baker et al., 2001a). In contrast, the mean minimum temperatures were warmer in both cities than the respective neighboring rural landscape, though the differential in Phoenix was greater.

The heat island intensity is also related to city size and population density (Oke, 1973; Brazel et al., 2000). For example, Baltimore's mean minimum temperature differential increased until the 1970s when the city population declined. Since 1970, the mean minimal temperature differential has leveled off. Phoenix also showed an increase in mean minimum temperature differential with an increase in population. However, because Phoenix has continued to grow explosively, the differential has continued to increase. In general, a non-linear relationship exists between mean minimum temperature differential and population density (Brazel et al., 2000).

3.2. Heat island effects

The differences in climate between city and countryside have biological and human implications. For example, as a result of climatic modification in temperate zone cities, leaf emergence and flowering times are earlier, and leaf drop later than in the surroundings (Sukopp, 1998; White et al., 2002). Additional effects of temperature on urban plants are presented in Section 5. The UHI affects human behaviors as well. UHI increases energy use, due to a greater demand for air conditioning, and increases the production of ground level ozone (Akbari et al., 1996, 2001). Ozone is created through a photochemical reaction of volatile organic compounds and NO_x in the presence of sunlight. If precursors are present, higher temperatures lead to greater ozone production in urban areas with direct consequences for human health (Akbari et al., 1996; Taha, 1997) and plant growth (Gregg et al., 2003; Section 5).

Differences in temperature between urban and rural areas also have been noted for soil. For woodland soils in the New York City metropolitan area, surface temperatures (2-cm depth) differed by as much as 3 °C between urban and rural forest patches (Pouyat et al., 2003); however, in Baltimore only a 0.5 °C difference on average was found between urban and rural forest soils (Savva et al., 2010). In the same study, the difference between urban turf grass and rural forest soils was approximately 3 °C in contrast to a difference of 6 °C in New York City.

3.3. *Atmospheric accumulation*

Certain atmospheric features are concentrated by urban areas. Precipitation is enhanced in and downwind of cities as a result of the higher concentrations of particulate condensation nuclei in urban atmospheres. Precipitation can be up to 5–10% higher in cities, and they can experience greater cloudiness and fog (Botkin and Beveridge, 1997). The probability of precipitation increases toward the end of the work week and on weekends due to a build up of particulates resulting from manufacturing and transportation (Collins et al., 2000). In contrast, wind velocities in cities are typically reduced due to increased surface roughness (Hough, 1995). Urban areas also concentrate materials from elsewhere. A carbon dioxide dome accumulates over cities in association with combustion of fossil fuels (Brazel et al., 2000). Anthropogenically produced forms of nitrogen are concentrated in and downwind of cities. Urban deposition is in the form of nitrate whereas agricultural nitrogen deposition is in the form of ammonium.

3.4. *Urban hydrology*

Urban hydrology is drastically modified compared to agricultural and wild lands (Paul and Meyer, 2001; Walsh et al., 2005a). Relativizing a water budget to 100 units of precipitation, and comparing urban to non-urban areas, evapotranspiration decreases from a 40% to 25%, surface runoff increases from 10% to 30%, and ground water decreases from 50% to 32% (Hough, 1995). Forty-three percent of precipitation exits the urban area via storm sewers, with 13% of that having first fallen on buildings. The hydrology in urban areas can be further modified by ecological structures. For example, reduced tree canopy cover in urban areas increases the flow rate of runoff and decreases the time lag between onset of storms and initiation of runoff (Hough, 1995).

Streams integrate human disturbances across catchments. Despite their critical role in transporting surface runoff, providing habitat for aquatic species, and attenuating nutrient pollution, the physical structure of streams can be severely impacted by urbanization, altering the transport of energy, materials, and certain ecosystem functions (e.g., Groffman et al., 2002; Elmore and Kaushal, 2008; Kaushal et al., 2008a). Increased runoff from impervious surfaces can contribute to stream channel incision, decreased ground water levels, and decreased interactions between ground water and "hot spots" of biogeochemical activity in organic-rich, upper soil horizons (Groffman et al., 2002). In addition, headwater streams have been converted to gutters, storm drains, and culverts (Elmore and Kaushal, 2008). For example, approximately 20% of stream miles are estimated to have been buried in the Gunpower-Patapsco watershed, a major tributary of the Chesapeake Bay in Baltimore, Maryland (Elmore and Kaushal, 2008).

Changes in the physical structure of suburban and urban headwater streams greatly amplify the effects of climate variability on downstream nutrient transport, propagate disturbances across stream networks, and contribute to cascading effects on coastal water quality (Kaushal et al., 2008a). Although much work has focused on conceptualizing and analyzing urban impacts on the physical structure, chemistry, and species diversity at the scale of stream reaches (Paul and Meyer, 2001; Walsh et al., 2005a), there has been less conceptual and empirical work focusing on longitudinal impacts along the urban river continuum. There is a need to link the impacts of headwater alteration due to urban land-use change with downstream transport of nutrients (e.g., Walsh et al., 2005b; Kaushal et al., 2006; Alexander et al., 2007) and assemblages of aquatic species (Meyer et al., 2007).

3.5. *Urban stream syndrome and urban stream restoration*

The "urban stream syndrome" is a conceptual model to describe the consistently observed ecological degradation of streams draining urban landscapes (Walsh et al., 2005a). This degradation includes elevated nutrients, increased organic and inorganic contaminants, increased hydrologic flashiness, and altered biotic assemblages (Walsh et al., 2007). In particular, streams draining urban landscapes have elevated concentrations and loads of nitrogen (Groffman et al., 2004; Wollheim et al., 2005; Kaushal et al., 2008a; Bernhardt et al., 2008) and phosphorus (Brett et al., 2005) compared to streams draining forested watersheds. In addition to elevated nutrients, urban streams can also have elevated concentrations of other contaminants such as salt from deicing of pavement (Kaushal et al., 2005b; Kaushal, 2009). For example, chloride concentrations increased 4-fold over approximately 25 years in streams draining into the drinking water supply of Baltimore, Maryland due to increasing coverage by impervious surfaces in the watershed, and an apparent legacy of salt storage in ground water (Kaushal et al., 2005b; Kaushal, 2009). Other studies in urban streams and storm drains have shown elevated concentrations of metals (Jang et al., 2005), organic contaminants (Scoggins et al., 2004), and bioavailable organic carbon and nitrogen (Seitzinger et al., 2002). Stream temperatures can also be elevated in urban areas due to the absence of riparian tree cover and drainage from heat-absorbing surfaces (Kinouchi et al., 2007; Nelson and Palmer, 2007).

The hydrology of urban streams has been greatly altered, leading to magnification of runoff during storm events, erosion, sediment transport, reconfiguration of stream channels, and alterations in the timing and amount of nutrient transport (e.g., Walsh et al., 2005a,b, Shields et al., 2008; Pizzuto et al., 2008). Much of the nature of urban hydrology is affected by the patterns of connectivity in the pathways of water flow. On the one hand, hydrologic connectivity is increased via storm drains between impervious surfaces and streams. On the other, hydrologic connectivity is decreased between streams and riparian zones as a result of channel incision. These contrasting changes in hydrologic connectivity may decrease the ability of streams to effectively process nutrients and to attenuate contaminant transport (Elmore and Kaushal, 2008; Kaushal et al., 2008b). Because of hydrologic alterations, the biological activity that can sequester both pollutants such as high levels of nitrate, and other contaminants such as metals, may not function effectively in urban areas. In general nutrients and contaminants are expected to be removed or retentained in hot spots in headwater areas, riparian zones, and floodplains. Large reductions in nutrient retention are therefore expected in urban watersheds during periods of intense precipitation (Cadenasso et al., 2008; Kaushal et al., 2008a).

The combined effects of increased contaminants, temperatures, and flashy hydrology can interact to change biological community and ecosystem responses in streams. For example, urbanization can degrade aquatic community structure and species diversity (Roy et al., 2005a,b; Morgan and Cushman, 2005). The declines in species diversity can be related to disturbance by flashy hydrology (Roy et al., 2005a) and changes in riparian vegetation cover (Roy et al., 2005b; Moore and Palmer, 2005). Although less work has investigated effects of the urban stream syndrome on ecosystem functions, there can be effects on biogeochemical processes (Meyer et al., 2005). Studies have also shown that urbanization can influence nutrient uptake processes at the stream reach scale (Meyer et al., 2005; Grimm et al., 2005; Mulholland et al., 2008). For example, uptake processes may be compromised by elevated nutrient concentrations and the saturation of biotic demand for nitrogen (e.g., Bernot and Dodds, 2005; Mulholland et al., 2008).

Because nutrient retention in streams may be compromised by hydrologic alterations and elevated nutrient concentrations, there is great interest in the ability of stream restoration and ecological engineering to improve nutrient retention in urban areas (e.g., Kaushal et al., 2008b; Craig et al., 2008; Roach et al., 2008). Hydrologic residence time appears to be a key factor related to N retention in both urban ground water and watersheds (Kaushal et al., 2008a,b). Thus, reducing runoff through improved stormwater management will be critical for improving water quality across a range of precipitation variability (Walsh et al., 2005b; Kaushal et al., 2008a). In addition, increasing organic carbon accumulation in stream sediments and fostering interaction between hydrologic flow paths and hot spots of denitrification may also improve N dynamics in restored streams (Groffman et al., 2005; Craig et al., 2008). Watershed restoration that encompasses both streams and their surrounding watersheds offers an opportunity to improve the form and function of stream systems beyond the stream reach scale and to integrate aquatic processes with urban landscape design (Cadenasso et al., 2008).

3.6. Streams as bioreactors/transformers of nutrients

Due to increased inputs of inorganic nutrients, in addition to alterations in temperature, stream habitat and biotic activity, many streams affected by urbanization may now be shifting from largely transportive systems reflecting anthropogenic changes in the landscape to transformers, in which the forms and ecological effects of nutrients and organic matter are actively changed (Kaushal and Lewis, 2005). For example, heterotrophic microbes in stream sediments can consume inorganic N and release organic N when labile organic carbon is abundant during baseflow conditions and when there is increased hydrologic residence time in streams (Kaushal and Lewis, 2005). Organic matter may represent a potentially important form of nutrients, contaminants, and energy subsidies in urban streams and more research on its origins and transformations across urban stream networks is necessary (Kaushal et al., 2005a).

The implications of urbanization for streams constitute a critical component of global change research (Grimm et al., 2008). Typically, concentrations and yields of inorganic nutrients such as nitrate, ammonium, and phosphate have been used as indicators of human impacts at the watershed scale (Groffman et al., 2004; Kaushal et al., 2008a). However, organic forms of nitrogen (N) and phosphorus (P) are high both in many streams draining minimally disturbed and in urban watersheds (e.g., Kaushal and Lewis, 2005; Kaushal et al., 2005a; Pellerin et al., 2006). These organic forms can be biologically available from a few days to several weeks (e.g., Seitzinger et al., 2002; Kaushal and Lewis, 2005; Kaushal et al., 2005a). Thus, organic N and P represent significant pools of nutrients capable of supporting stream ecosystem functions but are also capable of causing downstream eutrophication. However, little is known about factors regulating the generation, transport, and uptake of organic N and P in urban streams (Pellerin et al., 2006).

4. Substrate: urban soils

Soils are the foundation for many ecological processes such as biogeochemical cycling, the spatial distribution of plant assemblages and, ultimately, the location of human habitation and activity (Pouyat et al., 2007b). Although many soils in urban and urbanizing landscapes are modified by human activity, they serve, like unmodified soils, as both sources and sinks for nutrients and contaminants (Bullock and Gregory, 1991; De Kimpe and Morel, 2000; Pouyat et al., 2007a) and are a critical mediator of feedback mechanisms (Effland and Pouyat, 1997). As such, soils function in the urban landscape by retaining and supplying nutrients, serving as a growth medium and substrate for soil fauna and flora, and contributing to the hydrologic cycle through absorption, storage, and supply of water (Bullock and Gregory, 1991). In providing these services, soil plays a key role as the “brown infrastructure” of urban ecological systems, much in the same way urban vegetation, or green infrastructure, does (Benedict and McMahon, 2002).

Soil studies in urban areas have typically focused on highly disturbed and human-constructed soils along streets and in highly developed areas (e.g., Craul and Klein, 1980; Patterson et al., 1980; Short et al., 1986; Jim, 1993, 1998; Pouyat et al., 2007b). As a result urban soils have been viewed as drastically disturbed and of low fertility (Craul, 1999). The characteristics of soil, however, can vary greatly across the entire urban complex, including not only highly disturbed but also relatively undisturbed soils that are modified by urban environmental factors (Schleuβ et al., 1998; Pouyat et al., 2003).

4.1. Urban soil alterations

Urban factors directly and indirectly affect soil chemical, physical, and biological characteristics. Direct effects include physical disturbances, incorporation of anthropic materials, and burial or coverage of soil by fill and impervious surfaces (Craul, 1992; Jim, 1998; Schleuβ et al., 1998). The importance of fill is illustrated by the fact that the level of the substrate in old cities is generally higher than the surroundings (Sukopp, 1998). Soil management practices, e.g., fertilization and irrigation, which are introduced after the initial development, also are considered direct effects. The spatial pattern of these disturbances and management practices are largely the result of parcelization, or the subdivision of land as landscapes are developed for human settlement. The development of the landscape creates distinct parcels of disturbance and management regimes that will affect the characteristics of soil over time resulting in a mosaic of soil patches. Pouyat and Effland (1999) suggest that physical disturbances often lead to new soil parent material in which soil development occurs.

Indirect effects of urbanization on soils involve changes in the abiotic and biotic environment, which can influence the development of intact soils. Unlike direct physical disturbance, changes in environmental factors resulting from urbanization affect soil development at temporal scales in which natural soil formation processes are at work (Pouyat and Effland, 1999; Pickett and Cadenasso, 2009). Indirect effects include the urban heat island (Oke, 1995; Mount et al., 1999), soil hydrophobicity (White and McDonnell, 1988; Craul, 1992), introductions of exotic plant and animal species (Steinberg et al., 1997; Ehrenfeld et al., 2001), and atmospheric deposition of pollutants such as N (e.g., Lovett et al., 2000), heavy metals (Orsini et al., 1986; Boni et al., 1988;

Lee and Longhurst, 1992; De Miguel et al., 1997; Mielke, 1999), and potentially toxic organic chemicals (Wong et al., 2004).

4.2. Soil moisture

Soil-moisture regimes are an important factor of soil formation and function, and can be modified by urban environments. Disturbed soils in urban areas typically have limited infiltration capacities due to hydrophobic surfaces, crust formation, and soil compaction (Craul, 1992; Pouyat and Effland, 1999). Compounding slow infiltration rate is the potential of impervious surfaces to drain flow into pervious areas. This excessive water input can prevent infiltration even during small storm events. Soil-moisture levels also can be reduced in urban areas due to higher air temperatures, which generate higher evaporative and transpirative sinks. In contrast to factors that reduce soil-moisture levels, soils in urban areas often are irrigated and have abrupt textural and structural interfaces that can restrict drainage resulting in higher soil-moisture content in surface layers (Craul, 1992). Moreover, belowground infrastructures such as pressurized potable water distribution systems can leak as much as 20–30% of the water they carry into adjacent soils (Law et al., 2004).

Currently, we lack long-term continuous data with which to make comprehensive comparisons of soil moisture between urban and rural forest patches or between urban forest soils and disturbed soils. In the Baltimore Ecosystem Study (www.beslter.org), continuous measurements in a medium density residential area and adjacent deciduous woodland indicate high spatial variability (Fig. 2). Although variability exists within land-use type due to local patchiness, differences between forest and grass are much higher. Continuous measurements also allow understanding of the response of the system to extreme rainfall events. Such measurements show that precipitation has a more profound effect on the soil in the grass dominated area, presumably due to the absorptive properties of the leaf litter layer in the forest and the uneven distribution of water via stemflow. Synoptic measurements in the

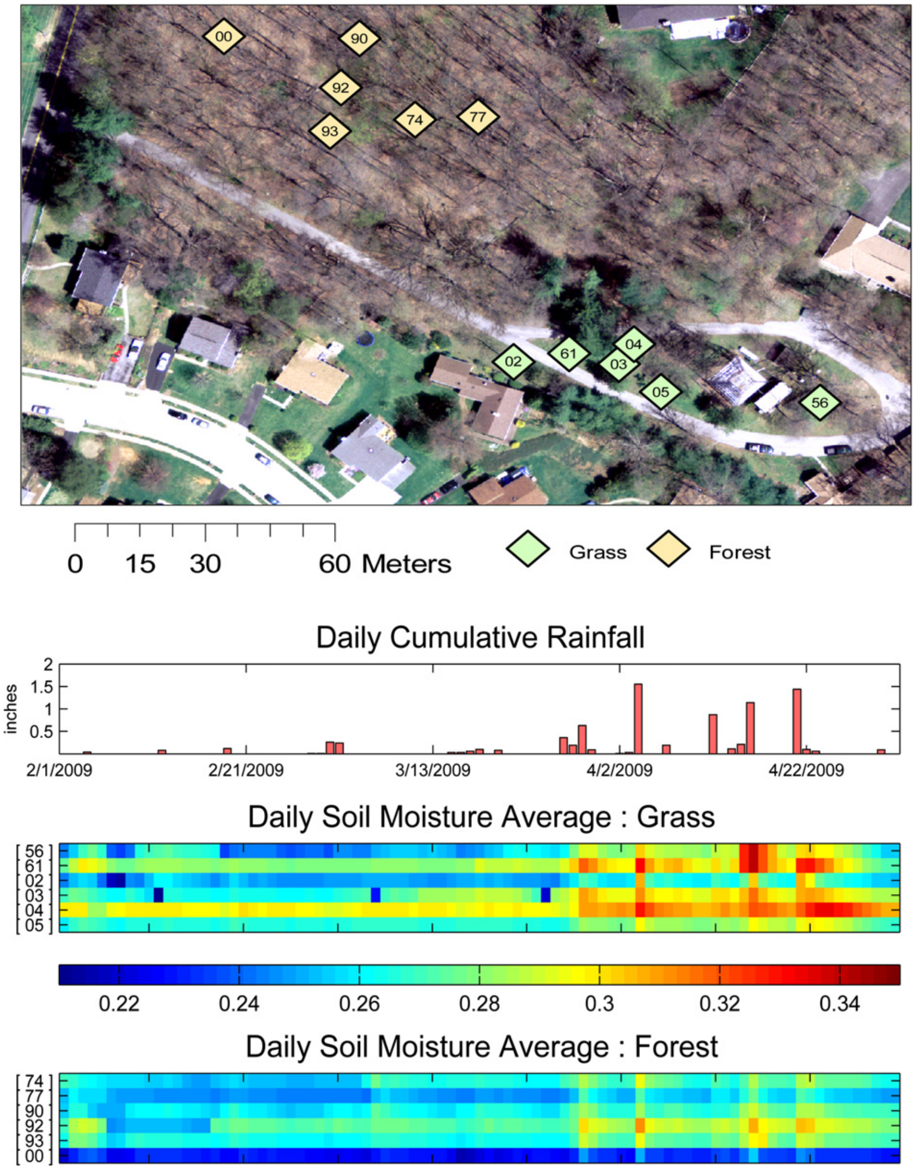

Fig. 2. Variability in soil moisture in contrasting urban habitats. Soil moisture to the depth of 5 cm as measured by a sensor array. The top panel is an aerial image of the sample area, with the location and designation of each sensor indicated. The second panel is the time course of rainfall over the measurement period. The two lowest panels show the soil moisture over time for each sensor location. The numbers in the left hand stub of each lower panel indicate the sensor. Measurements were taken at 20 min intervals.

same area indicate that differences in soil moisture occur between land-use and -cover types in urban landscapes. Residential lawns within 150-m of a forest patch showed that unirrigated turf soils had higher moisture levels than forest soils during the summer. There were no differences between the two patch types at the end of October after leaf drop. Presumably, differences in the summer were due to higher transpiration rates of the broad-leaved trees. In late fall and early winter, the moisture level in turf soil increased more rapidly than in forest soil, possibly reflecting concentrated runoff from roof gutters or snow melt. How these differences affect C and N cycling is an open question. However, knowledge of heterogeneity in both soil moisture and temperature is important in modeling urban water and nutrient cycles.

4.3. *Soil contamination*

Studies of forest remnants along urbanization gradients suggest that urban remnants can receive high amounts of heavy metals, organic compounds, and acidic compounds via wet and dry atmospheric deposition (Pouyat et al., 2007a,b). Lovett et al. (2000) quantified atmospheric N inputs in oak forest stands along an urbanization gradient in the New York City metropolitan area. They found that urban forests received 50–100% greater N fluxes in throughfall than rural forests. The input of N declined rapidly in the suburban stands 45 km from Manhattan. It was proposed that dust from New York City, rich in alkaline particles of Ca^{2+} and Mg^{2+}, act like an “urban scrubber,” removing acidic gases such as NO_3^- and SO_4^{2-} from the atmosphere and depositing the products on the city as coarse particulates. Despite the urban scrubber effect, most of the dry deposition of nitrate was from gaseous nitrogen oxides, which were in much higher concentration in the city than in rural sites. It was hypothesized that the alkaline urban dust originated from building materials and construction activity within the city (Lovett et al., 2000). Similar results were found in Louisville, KY, US and the San Bernardino Mountains in metropolitan Los Angeles where both N and base cation deposition rates into urban forest patches were higher than in rural forest patches (Bytnerowicz et al., 1999; Carreiro et al., 2009; Fenn and Bytnerowicz, 1993).

Evidence of similar depositional patterns was found in the form of heavy metal content in soils along urbanization gradients in the New York City, Baltimore, and Budapest Hungary metropolitan areas. Pouyat et al. (2008b) found up to a 2- to 3-fold increase in contents of lead (Pb), copper (Cu), and nickel (Ni) in urban than in suburban and rural forest remnants for these cities. A similar pattern but with greater differences was found by Inman and Parker (1978) in the Chicago, IL, US, metropolitan area, where levels of heavy metals were more than five times higher in urban than in rural forest patches. Other urbanization gradients have shown the same patterns, though for smaller cities differences between urban and rural stands were narrower (Carreiro et al., 2009; Pavao-Zuckerman, 2003). Besides heavy metal and base cation deposition, Wong et al. (2004) found a steep gradient of Polycyclic Aromatic Hydrocarbon (PAH) concentrations in forest soils in the Toronto metropolitan area, with concentrations decreasing with distance from the urban center to surrounding rural areas by a factor of 60 or more.

For urban soils, elevated heavy metal concentrations are almost universally reported, often with high variances (Thornton, 1991; Li et al., 2004; Wong et al., 2006). Most of the heavy metal sources in cities have been associated with roadside environments (Van Bohemen and Janssen van de Laak, 2003; Zhang, 2006), interior and exterior paint (Trippler et al., 1988), refuse incinerators (Walsh et al., 2001), industrial stack emissions (Govil et al., 2001; Kaminski and Landsberger, 2000), management (Russell-Anelli, 1998) and industrial waste (Schuhmacher et al., 1997). In Baltimore City, Pouyat et al. (2007b) found that without accounting for background levels of parent material, soil concentrations (mg kg^{-1}) of 12 heavy metals were highly variable and generally did not correspond to land use and cover. However, Yesilonis et al. (2008) found that metal levels were indirectly related to distance from major roads and age of housing stock after accounting for background levels using the same dataset (Fig. 3).

Field investigations addressing the relative importance of natural vs. anthropogenic factors in the formation of soils in urban landscapes has been suggested as a novel area of research in the soil sciences (Galbraith et al., 1999; Pouyat and Effland, 1999; Pickett and Cadenasso, 2009). This is particularly true in the separation of the underlying urban and non-urban factors associated with studies of urbanization gradients and urban landscape mosaics (Pouyat et al., 2008b).

Even in highly modified landscapes, characteristics of the native parent material persist in urban soils and urban environmental effects on native soils can occur at distances beyond the boundaries of cities. Pouyat et al. (2007b) compared forest patches along urbanization gradients in the Baltimore, New York, and Budapest metropolitan areas to investigate the relative importance of urban environmental factors vs. parent material on soil chemistry. These metropolitan areas differ in population densities, surface areas, surface geologies, and transportation systems. Forest soils responded to urbanization gradients in all three cities, though different spatial pattern of development, pollution sources, parent material, and site history influenced the soil chemical response of each city. Pouyat et al. (2008b) suggest that the changes measured resulted from locally derived atmospheric pollution of Pb, Cu, and to a lesser extent Ca, which was more extreme at the urban end of the gradient, but extended beyond the political boundary of each city. Moreover, in Baltimore and Budapest the soil chemical response and analysis of soil pollution effects were confounded by differences in parent material found along those urbanization gradients (Pouyat et al., 2007a,b; Yesilonis et al., 2008; Bityukova et al., 2000; Facchinelli et al., 2001).

4.4. *Soil C and N dynamics*

Soil modifications resulting from urbanization can modify soil C and N dynamics. To assess the potential effect on soil organic C, data from 1 m deep profiles of made soils from five different cities, and from the surface 0–15 cm of several land-use types in Baltimore were analyzed (Pouyat et al., 2002, 2006). Soil pedons from five cities showed the highest soil organic C densities in loamy fill (28.5 kg m^{-2}) with the lowest in clean fill and old dredge materials (1.4 and 6.9 kg m^{-2}, respectively). Soil organic C for residential areas (15.5 ± 1.2 kg m^{-2}) was consistent across cities. A comparison of land-use types showed that low density residential land had 44% higher and institutional land had 38% higher organic C densities than commercial land. Therefore, made soils, with their physical disturbances and inputs of materials by humans, can greatly alter C storage in urban systems (Pouyat et al., 2008a).

The various studies of soil C and N dynamics of unmanaged urban forest stands, highly disturbed soils, and surface soils of various urban land-use types demonstrated that urbanization can directly and indirectly affect soil C pools and N-transformation rates. Soil C storage in urban ecosystems is highly variable (Pouyat et al., 2006, 2009). How generalizable these results are across cities located in similar and dissimilar life zones needs to be investigated. In addition, more data are needed on highly disturbed soils, such as land-fill, managed lawns, and covered soils to make regional and global estimates of soil C storage and N-transformation rates in urban ecosystems. Specific uncertainties include the quality of the C inputs governed by the presence of exotic plant species litter and

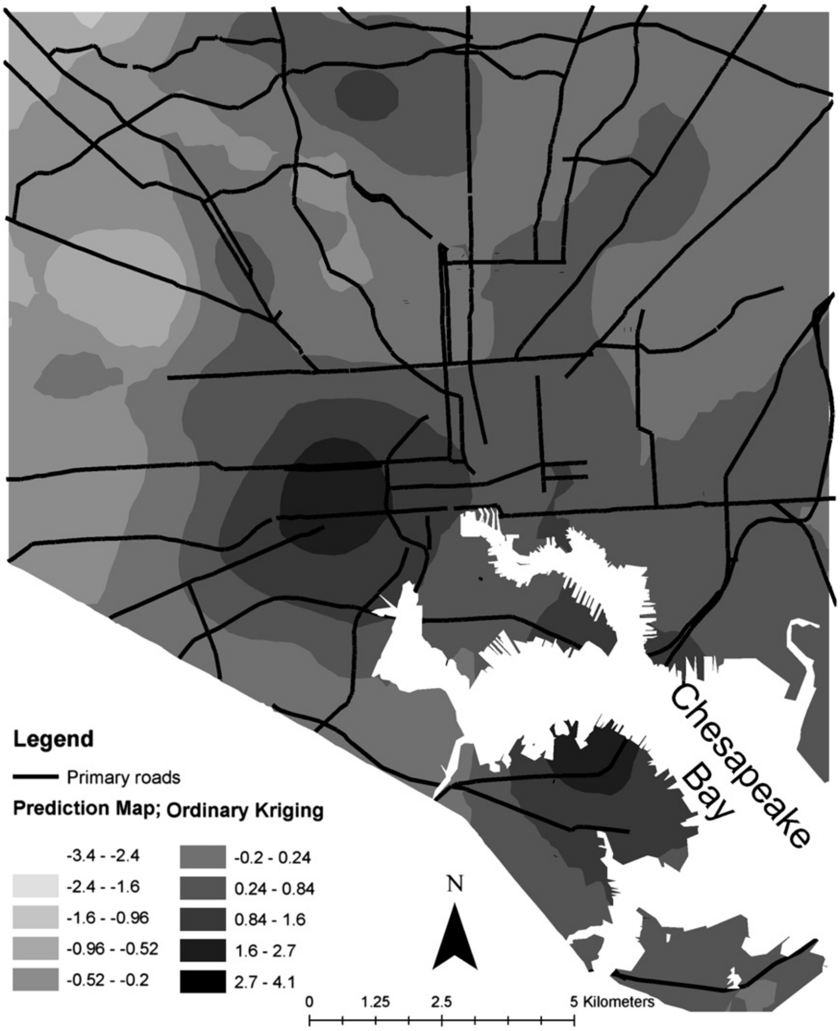

Fig. 3. Concentration of soil borne heavy metals, Cu, Pb, Zn in kg ha^{-1} after citywide variation in surface geology has been accounted for. The metals are represented by the second principal components axis, the values of which are analyzed via ordinary kriging and displayed in relationship to the major roads in Baltimore City, MD. Reproduced with permission from Yesilonis et al. (2008).

by stress effects on native species litter, the fate of soil C in covered soils, measurements of soil C densities at depths greater than 1 m particularly in "made" soils, the effects of specific management inputs on N-transformation rates, and spatially delineating disturbed and made soil types (Pouyat et al., 2003).

5. Organisms: urban vegetation

The study of vegetation in urban areas has a long history. For example, in Europe, studies by De Rudder and Linke (1940) documented the flora and fauna of cities during the early decades of the 20th century. After World War II, studies examined vegetation dynamics of bombsites in cities (e.g., Salisbury, 1943). At the same time, ecologists in the US focused on describing flora in areas of cities minimally altered by humans, such as parks and cemeteries. One of the first comprehensive studies of urban vegetation and environments was conducted by Schmid (1975) in Chicago.

Forests in urban stands tend to have lower stem densities, unless those stands are native remnants in large parks or former estates (Lawrence, 1995). Street trees and residential trees tend to be larger than those in forest preserves, natural areas, and wild lands in the Chicago region (McPherson et al., 1997), though individual trees in urban sites are often stressed, especially on or near streets (Ballach et al., 1998). Street trees in Chicago accounted for 24% of the total Leaf Area Index (LAI) and 43.7% of the LAI in residential areas (Nowak, 1994). Compared to the nearly continuous pre-urban forest cover in mesic forest regions of the US, tree cover of cities is approximately 31%. In contrast, for prairie-savanna and desert regions, tree canopy cover has increased over pre-urban levels (Nowak et al., 1996). In forest biomes, forest cover decreases as a proportion of total land area from non-metropolitan to central city counties, with a step reduction between non-metropolitan and peripheral counties (Brooks and Rowntree, 1984). Local effects on forests also occur. For example, forest patches adjacent to residential areas experience increased edge openness (Moran, 1984) and a retreat of their margins due to recreational use, especially by children, and damage to regenerating plants (Bagnall, 1979). Such regeneration failure is frequent in urban and suburban stands, due to reduced natural disturbance or gap formation, substitution of unfavorable anthropogenic disturbances such as frequent ground fires and trampling, and competition with exotics (Guilden et al., 1990).

The composition of urban and suburban forests differs from wild and rural stands. Species richness has increased in urban forests as a whole, but this is due to presence of exotics (Zipperer et al., 1997). The presence of exotics is often notable in urban vegetation. The percentage of the flora represented by native species decreases from urban fringes to center city (Kowarik, 1990). The same is true even within forest fragments from rural-to-urban contexts (Duguay et al., 2007). Along the New York City urban-to-rural gradient, the number of exotics in the seedling and sapling size classes is greater in urban and suburban oak stands (Rudnicky and McDonnell, 1989). A more detailed study of the vegetation of the forest stands along this transect indicated that the composition of the canopy did not differ from urban-to-rural sites. However, the structure of the forest as expressed in saplings, seedlings, vines, and herbs did differ across the transect, with notable breaks in character between rural forests and those closer to New York City (Cadenasso et al., 2007b). In the US urban flora in general, the representation of exotics increases with time since statehood, correcting for human population density, indicating a role of time of exposure to invasion (McKinney, 2001). Rapoport (1993) found the number of noncultivated species to decrease from fringe toward urban centers in several Latin American cities. In Mexico City, there was a linear decrease in the number of species per ha from 30–80 encountered in suburbs to 3–10 encountered in the city center. At a given housing density, more affluent neighborhoods had more exotic species than less affluent ones in Bariloche, Argentina. In Villa Alicura, Argentina, exotic species increased with increasing local site alteration by humans. Near homes there were 74% exotics, while there were 48% along river banks. Although exotics were present along all roads, the number decreased on roads less frequently used. No exotic species were found outside the town. Paths in rural recreation areas (Rapoport, 1993) and in urban parks (Drayton and Primack, 1996) have enhanced the presence of exotics. In an urban park in Boston, of the plant species present in 1894, 155 were absent by 1993, amounting to a decrease from 84% to 74% native flora. Sixty-four species were new. In addition to trails, Drayton and Primack (1996) relate fire and trampling to the change in exotics.

The ecology of the entire city as a system is represented by research relating species richness to the characteristics of cities. For instance, the number of plant species in urban areas correlates with the human population size. Species number increases with log number of human inhabitants, and that relationship is stronger than the correlation with city area (Klotz, 1990). Small towns have from 530 to 560 species, while cities having 100,000–200,000 inhabitants have upwards of 1000 species (Sukopp, 1998). Age of the city also affects the species richness, with large, older cities exhibiting more plant species than younger cities (Sukopp, 1998; Kowarik, 1990). These assemblages are characteristic in Europe, with 15% of species shared among cities (Sukopp, 1998).

Urban areas show a preponderance of trees of wetland or floodplain provenance, due to the lower oxygen tensions shared by wetlands and impervious urban soils (Spirn, 1984). Even when the overstory remains similar, the herbaceous flora of urban forests is likely to contrast with rural or wild stands (Wittig, 1998; Cadenasso et al., 2007b). In Japan and Minnesota, US vascular plant diversity increases with the area of the stand (Iida and Nakashizuka, 1995; Hobbs, 1988). In some urban stands, the adjacent land use affects species composition. The interior of residential forests had more exotics than forests abutting either roads or agricultural zones (Moran, 1984).

Compositional changes are not the only dynamics in urban biota. In addition, phenology has changed. Flowering in European cities now begins 4–17 days earlier than in rural areas (Roetzer et al., 2000). The growing season length has increased on average approximately 8 days, primarily due to earlier bud break. Such phenological changes are responses to the urban heat island (Section 3.2). In contrast, because dormancy is signaled by photoperiod, the end of the season for most plants was not affected (White et al., 2002). Spring blooming plants flowered up to 9 days earlier in the city than in the countryside (Neil and Wu, 2006), and in particular, ephemeral annuals, early spring flowering plants, and insect pollinated plants were more sensitive to environmental changes than perennials or plants that bloom late in the growing season (Neil and Wu, 2006). Plants and animals in cities have evolved compared to populations outside of cities (Sukopp, 1998). The famous population genetic differentiation in *Agrostis tenuis* and *Plantago lanceolata* in lead contaminated urban sites is an example (Velguth and White, 1998).

5.1. Urban vegetation heterogeneity

A major feature of urban vegetation is its spatial heterogeneity, created by the vast array of building densities and types, different land uses, and different social contexts. To characterize this heterogeneity, plant habitats have been classified using various systems. Stearns (1971), one of the first American ecologists to call for research in urban landscapes, identified three major habitat types—ruderal, managed, and residual. Building upon this foundation, Brady et al. (1979) proposed a continuum of habitats from the natural to the highly artificial. Dorney (1977), using a similar approach, proposed an urban–rural continuum from a planning perspective and identified six representative land zones—central business district, old subdivisions, new subdivisions, urban construction zones, urban fringe, and rural. Each zone was characterized by three components or subsystems: cultural history, abiotic characteristics, and biotic features.

In addition to assessing spatial heterogeneity, classification systems recognize the importance of characterizing natural habitats in urban landscapes. To assess urban wildlife habitat in Germany, Sukopp and Weiler (1988) developed a biotope mapping and conservation strategy based on phytosociologic-floristic and faunal characteristics. Rogers and Rowntree (1988) developed a system to classify vegetation using life forms. The process was used to assess natural resources in New York City (Sissini and Emmerich, 1995). Based on site histories, Zipperer et al. (1997) classified tree-covered habitats as planted, reforested, and remnant. These approaches allow for spatially-explicit comparisons between vegetation and other variables. More recent classification schemes recognize the need to integrate vegetation with the built and surface characteristics of urban areas (Cadenasso et al., 2007a).

The vegetation and floristic studies of nature in cities share key characteristics. They are largely descriptive, but illustrate spatial heterogeneity as a source of diversity, and suggest a functional role for landscape structure (Rebele, 1994). While it is legitimate to view the city as an open and dynamic ecosystem, few of the plant ecological studies document successional processes (Matlack, 1997) or expose the functional relationships of the vegetation (Mucina, 1990). More recent experiments have documented the ability of contrasting environments along an urban-to-rural gradient, acting on a common experimental seed bank, to cause the rate of succession to change (George et al., 2009). Over 5 years 2.1 °C greater temperature and 20% greater CO_2 vapor pressure deficit were associated with 2–3 times greater biomass, and significantly greater ratio of perennials to annuals in the urban sites.

6. Organisms: urban animals

The long history of research on the animal life of cities has been documented by Douglas (1983), Nilon and Pais (1997) Marzluff et al. (2001), McKinney (2002), Chace and Walsh (2006), and Adams et al. (2006) for, Australian, European and North American cities. The majority of the studies reviewed previously (Pickett et al., 2001) was apparently designed to document the fauna or to answer specific management questions. Few studies focused on processes identified in mainstream ecology as major foci. Over the past decade, however, considerable advances have been made in addressing processes like extinction-colonization, competition, trophic dynamics, and behavioral adaptation (Shochat et al., 2006). Strides have also been made in addressing socio-economic drivers influencing animal community composition (Hope et al., 2004; Kinzig et al., 2005; Melles et al., 2003; Nilon and Huckstep, 1998). We review the patterns that have emerged from the history of descriptive studies of urban animal communities, then highlight the advances made in understanding the processes underlying these patterns. We begin with coarse scale or gradient comparisons, followed by patch-oriented studies. Finally, we review the burgeoning study of human–wildlife interactions. These interactions are complex, with some portions of human communities simultaneously having either positive or negative interactions with wildlife (DeStefano and DeGraaf, 2003). As the world citizenry becomes increasingly urban, human understanding of wildlife is increasingly shaped by urban wildlife encounters, with profound consequences for natural resource policy (Turner et al., 2007; DeStefano and DeGraaf, 2003).

Birds, mammals and terrestrial invertebrates continue to be the best studied taxonomic groups, with aquatic fauna, reptiles and amphibians less studied, despite a long time recognition of this imbalance (Luniak and Pisarski, 1994). In addition, the fauna of greenspaces are relatively well studied, with built up and derelict areas and water bodies less well known, though some landmark studies have demonstrated the benefits of focusing research on developed lands (Turner et al., 2004). Andrzejewski et al. (1978) and Klausnitzer and Richter (1983) described urban-to-rural gradients defined by human population density and building density that impacted the occurrence and abundance of species. Other commonly used indices of urbanization include the amount of impervious surface, road density or time since development (Blair, 1996; McKinney, 2002). Across these gradients, diversity declines in nearly all taxa studied (McKinney, 2002), with some taxa, e.g., amphibians, exhibiting steeper declines than others (Clergeau et al., 2006). However, in Palo Alto, CA, species richness of butterflies and birds was greatest at sites intermediate between nature reserves and dense urban areas (Blair, 1999). In taxa that do exhibit decline from rural-to-urban sites, patterns suggest dispersal or degree of specialization to be among the causes. Analysis of several soil arthropod taxa in Warsaw showed that the degree of decline depended on the size of the regional species pool. More diverse taxa, such as carabid beetles and spiders, lost a higher proportion of the original species, while most species in less diverse groups, such as isopods and daddy longlegs, were still present in the city. One possible explanation is that species rich groups likely have many specialized species that cannot survive in the urban setting. Thus, composition of urban animal communities tends to shift toward generalist species, often those with high reproductive capacities (DeStefano and DeGraaf, 2003). These generalists typically occur at higher densities in urban than in nearby wild land or rural habitats (McKinney, 2002; Shochat et al., 2006). It is possible to identify some specific features of successful generalists. For example, Møller (2009) hypothesized that widely distributed species, with high dispersal abilities, and possessing a high rate of innovation, a high level of risk taking, and a rapid life history would be selected for in urban habitats. The spatial patterns of generalists and specialists (Fig. 4) result in a peak avian diversity at intermediate levels of urbanization, where fragmented wild land or native lands are interspersed with lower density residential development (McKinney, 2002; Marzluff and Rodewald, 2008). Consensus on this pattern, however, is lacking, perhaps due to differences among studies in the resolution at which urban gradients are measured.

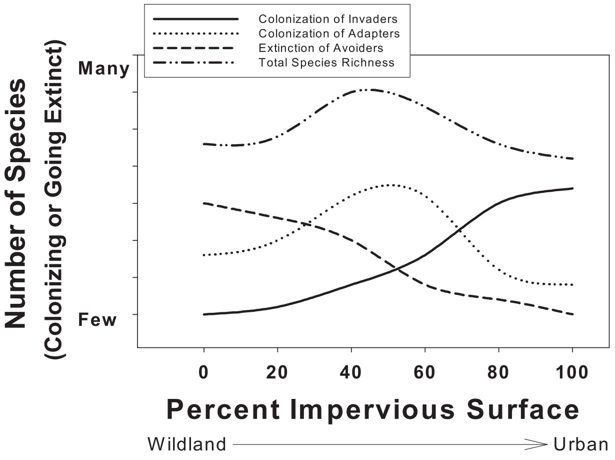

Fig. 4. Responses of species with different degrees of tolerance and adaptation to human influence along a gradient of urbanization, represented by imperviousness. Used with permission from Marzluff and Rodewald (2008).

Patch-oriented approaches are particularly well developed in studies of mammals. Studies in Australia, Europe, and North America have documented the importance of patch configuration, patch disturbance, and land use and cover in areas surrounding habitat patches on mammal populations and communities (Nilon and Pais, 1997; Adams et al., 2006). Recent studies have expanded patch approaches to process-oriented approaches (Parker and Nilon, 2008).

The dominance of urban communities by exotics and the large degree of similarity in species composition among cities have together been termed biotic homogenization (Blair, 1999; McKinney and Lockwood, 1999; Lockwood et al., 2000; McKinney, 2005). It is thought to arise from 1) similar patterns of colonizing and declining species owing to common characteristics of urban environments regardless of the surrounding biome, and 2) transportation of similar subsets of species across geographical boundaries (McKinney and Lockwood, 1999; McKinney, 2006). Several mechanisms account for the increased success of non-native species. Factors invoked as supporting homogenization include: the prevalence of disturbed habitats in cities promoting exotic species invasions, and common features of the physical environment of cities, such as heated buildings, irrigated and limed lawns, and increased air temperature. Biotic homogenization has been shown for birds (Lockwood et al., 2000) and plants (McKinney, 2004; Schwartz et al., 2006). For invertebrates, similarity of faunas among distant cities may not be the rule despite standardized sampling techniques (Niemelä et al., 2002). Analysis of soil macroinvertebrates in European cities showed high degrees of homogenization for earthworms, but less for isopods and millipedes (Korsós et al., 2002). Response of carabid beetle assemblages to urbanization varied locally, and due to the small size of these animals compared to vertebrate species, residual urban forest patches contained the same species as rural forests in some cities (Niemelä et al., 2002). Furthermore, native invertebrate species that can survive in the urban landscape come from zoogeographically distinct species pools, and do not spread beyond those boundaries unless humans transport them. The presence of these native “urban exploiters” (Blair, 2001) acts against biotic homogenization. These species are often close taxonomic relatives and ecologically equivalent, thus can be considered “urban vicariants”.

As the patterns of diversity and abundance associated with urbanization have crystallized, competing explanations have emerged. Application of island biogeography theory finds, in general, that species–area relationships are preserved in urban patches (Fernández-Juricic and Jokimaki, 2001; Donnelly and Marzluff, 2004; Wolf and Gibbs, 2004). In some studies, patch size influenced species composition rather than species richness, with organisms at higher trophic levels being those lost at smaller patch sizes (Gibb and Hochuli, 2002). A few studies have directly quantified the extinction and colonization processes that the theory proposes as underlying patterns of diversity globally (Crooks et al., 2001; Leston and Rodewald, 2006). According to this view, the species roster in a patch is the combination of the species colonizing the novel habitats formed with urbanization and those remaining after local extinctions are caused by isolation or habitat alteration from urbanization. Variation in habitat associations and dispersal distances are thought to be the primary drivers of the extinction and colonization rates for different species (Marzluff and Rodewald, 2008), though behavioral mechanisms have also been invoked, such as settlement biases in migratory bird species or decision rules in response to pressures like nest predation (Leston and Rodewald, 2006; Borgmann and Rodewald, 2004). One prediction of this view is that urban habitats are not always lower in diversity than non-urban ones. Rather, diversity depends on the sum of extinction and colonization rates, which differ regionally and taxonomically. At moderate levels of urbanization, species richness may actually be higher than in nearby wild lands.

An alternative, though not mutually exclusive view, is that species interactions, such as competition, play a key role in shaping community composition in urban habitats. For example, altered habitat structure and resource availability may favor colonizing species over native ones in competition for resources, leading to local extinctions of native species and elevated densities of a few strong competitors. Proponents of this view argue further that resource availability is elevated and more consistently available in urban environments, providing the opportunity for individuals in poorer condition to persist longer than in nearby wild land environments or less urbanized patches. Several predictions have been derived from this theory: a) that evenness is reduced along with diversity in urban habitats, b) that the most dominant competitors in urban environments are also the most efficient foragers, and c) within species, that the distribution of body condition among individuals in urban populations should be strongly skewed toward lower quality individuals (Shochat, 2004).

Finally management of the urban landscape can play an important role in shaping urban community structure, especially for smaller organisms. Attempts to selectively eliminate pests, actively encourage certain organisms, and accidentally transporting species with soil and horticultural plants, can modify species composition and abundance.

6.1. *Trophic dynamics*

Consistent patterns of human influence on urban food webs and trophic dynamics are emerging (Warren et al., 2006b). In terrestrial systems, strong bottom-up influences are frequently recognized (Faeth et al., 2005). Humans commonly subsidize resources, generating elevated productivity, particularly in cities in dry land environments (Shochat et al., 2006). Elevated densities for species of birds, arthropods, and mammals have all been associated with these human subsidies (DeStefano and DeGraaf, 2003). The literature is divided on whether the elevated densities constitute resource matching (Rodewald and Shustack, 2008a,b) or over matching (Shochat, 2004). It is noteworthy that aquatic systems, by contrast, often experience reduced basal resources, with frequently impoverished lower trophic levels in urban streams (Overmyer et al., 2005).

Top-down effects in urban food webs present a more complex picture. Predator diversity is often but not always decreased, and middle trophic levels may experience reduced predation (Bowers and Breland, 1996; Shochat et al., 2004; Parker and Nilon, 2008). An array of top predator species, from coyotes to cooper's hawks are finding ways to exploit more densely urbanized areas than was once thought possible (Morey et al., 2007; Gompper, 2002; Roth and Lima, 2003). In addition, human subsidies are thought to lead to artificially elevated densities for some predators, like cats and corvid birds, though this does not always result in elevated predation rates (Marzluff et al., 2001; Patten and Bolger, 2003; Rodewald and Shustack, 2008b). Humans often act directly as top predators, especially in landscaped and built environments, where chemical pesticides eliminate particular herbivore or detritivore species. But humans can also promote other arthropod groups, like lacewings, ladybugs, or soil nematodes, thought to be beneficial in controlling pests. Reticulate food web structure and predator compensation make top-down and bottom-up controls difficult to predict from knowing the composition of predator and prey communities (Patten and Bolger, 2003).

As in the broader field of ecology, "brown" food webs, i.e. detritus based or decomposer food webs, present a new frontier. The basis of decomposer food webs, plant detritus, is often spatially rearranged especially in suburban areas (e.g., Byrne et al., 2008). While urban forest patches retain their forest floor, thus maintaining a complex soil food web, leaf litter is removed in residents' yards and parks, and concentrated in compost piles. More specialized leaf litter feeding fauna will be absent in these habitats. Generalized detritivores will still survive because soil is amended with organic matter in the form of topsoil, finished compost or mulch. Nevertheless, decomposer food webs in urban forests appear to be as complex as food webs of the more well-described macro-fauna, with forest floor invertebrates experiencing seasonally varying degrees of top-down control by salamander predators (Walton, 2005; Walton et al., 2006).

Behavioral mechanisms play important roles in the adaptation or exclusion of animal species from urban and suburban environments. Anthropogenic noise leads to shifts in pitch and loudness of bird songs and shifts in calling behavior in some frogs (Slabbekoorn and Peet, 2003; Brumm and Todt, 2002; Sun and Narins, 2005; Warren et al., 2006a). However, species without the flexibility to modify their signals may be at a disadvantage in urban settings (Slabbekoorn and Peet, 2003). Synurban species, those that proliferate in urban settings, appear to possess a suite of common behavioral characteristics, including reduced wariness, increased intraspecific aggression, and increased tolerance of human presence (Gliwicz et al., 1994; Ditchkoff et al., 2006). This suite of traits is called an "urban wildlife syndrome" (Gliwicz et al., 1994; Parker and Nilon, 2008). Many of the altered behaviors observed in urban animals are density dependent, suggesting a direct connection from human resource subsidies to these behavioral shifts (Parker and Nilon, 2008). Other behavioral shifts in urban–suburban animals include altered seasonality, such as denning and hibernation in black bears (Beckmann and Berger, 2003). Whether synurbanization and other behavioral shifts are a consequence of phenotypic plasticity, pre-adaptation, or evolutionary adaptation is an unresolved question.

The retention of substantial pools of species diversity in cities presents an opportunity for an increasingly urban citizenry to encounter "nearby nature" (Kaplan and Kaplan, 1990). Humans appear to notice both diversity and its absence (Fuller et al., 2007) as well as to actively foster certain species (DeStefano and DeGraaf, 2003). Access to diversity of species and habitats, however, is not equitably distributed across social groups, according to a growing body of studies (Nilon and Huckstep, 1998; Hope et al., 2004; Kinzig et al., 2005; Melles et al., 2003).

Valuation of species is complex (Kellert, 1984), with contradictory viewpoints frequently expressed about the same species (e.g., mountain lions in Los Angeles (Davis, 1998). Wildlife managers consider regulation of populations of “overabundant” species, those exceeding cultural carrying capacities, to be more difficult in densely populated areas due to constraints placed on options for management, in particular lethal management techniques (DeStefano and DeGraaf, 2003; Adams et al., 2006; Perry and Perry, 2008). Delays in application of population controls may lead to rapid increases in pest or invasive species populations beyond sizes that are feasible to manage (Perry and Perry, 2008). Much current research in wildlife and pest management is aimed at addressing the particular problems of working in human-populated areas.

7. Interaction of the substrate and organism factors: urban biogeochemistry

Having now reviewed the substrate and organism factors separately, their interactions can be explored. These interactions, in part, take the form of biogeochemical fluxes. This central topic of ecosystem ecology has become a key aspect of urban ecology. Biogeochemistry in urban systems is an extremely broad area, which can be divided into concerns of 1) nitrogen and carbon budgets and retention, 2) urban footprints, 3) pollution effects on nutrients, 4) urban convergence as illustrated by lawns, and 6) the role of non-native invasive species in biogeochemistry.

7.1. Budgets of nitrogen and carbon: processing, retention, and sequestration

One of the earliest modern ecological approaches to urban systems was the assessment of biogeochemical budgets of whole cities (Boyden, 1979; Odum and Odum, 1980). Contemporary ecology propounds a systems view that goes substantially beyond the simple budgetary approach (Jones and Lawton, 1995). Although some ecosystem studies continue to apply the laws of conservation of matter and energy to generate budgets for ecosystems as though systems were “opaque boxes”, the structural details and richness of interacting processes that take place in the box are a major concern of contemporary ecosystem ecology. In addition, the assumptions of closed system boundaries and homeostasis that were associated with the ecosystem theory used by many early geographers have been replaced (Zimmerer, 1994). There is a new theory of ecosystems (e.g., Pickett et al., 1994) that was not available to those who initiated the budgetary approach to urban systems (e.g., Boyden, 1979). The basic concept of the ecosystem can serve the various disciplines (Rebele, 1994) but the internal heterogeneity of the systems and the processes within them are of substantial contemporary interest.

Recent biogeochemical budgets of whole cities or of urban watersheds as ecosystems are exemplified by questions about nitrogen retention, a key ecosystem service relevant to water and air quality. These budgets compare nitrogen inputs from fertilizer, atmospheric deposition, food for humans and pets and manufactured materials with outputs in streamwater, sewage, harvest or export of agricultural or manufactured products, and gaseous losses (Kaye et al., 2006). They are a strong platform for evaluating how increased nitrogen inputs associated with human activities contribute to water and air pollution. Soluble nitrogen exports contribute to eutrophication of water, especially coastal salt water, and a variety of nitrogen gases contribute to degrading air quality through either the greenhouse effect, tropospheric ozone pollution, or stratospheric ozone destruction (Galloway et al., 2003).

While one might expect urban ecosystems to have limited abilities to retain nitrogen given the extensive modification of soils and hydrology (Sections 3 and 4), recent urban ecosystem nitrogen budgets have shown surprisingly high retention. Baker et al. (2001b) compiled a comprehensive budget for the Central Arizona Phoenix (CAP) metropolitan region and found that inputs of N in fertilizer, food and atmospheric deposition greatly exceeded exports in streamwater and harvest. They concluded that the apparent retention consisted of gaseous losses and storage in soil. The storage in soil is supported by other analyses showing accumulation of inorganic and organic N in CAP soils (Zhu et al., 2006). The analysis by Baker et al. (2001b) highlighted the importance of considering unique urban nitrogen fluxes, such as those associated with pet food and waste.

Groffman et al. (2004) computed a nitrogen budget for an 81 ha suburban Baltimore watershed having 22% impervious surface by comparing inputs from atmospheric deposition and lawn fertilizer with streamwater outputs. Their finding of greater than 70% retention over a 3 year period, along with other studies showing high retention, motivated a search for just where nitrogen is being retained in urban and suburban watersheds. Locations hypothesized to be important for retention include lawns (Raciti et al., 2008), riparian zones and wetlands (Groffman et al., 2002, 2003; Groffman and Crawford, 2003; Kaushal et al., 2008a; Zhu et al., 2004), and streams (Groffman et al., 2005; Hale and Groffman, 2006; Grimm et al., 2005; Gibson and Meyer, 2007; Arango and Tank, 2008; Mulholland et al., 2008; Kaushal et al., 2008b; Klocker et al., 2009). Using budgets as a tool for driving watershed ecosystem research and management programs is a long established idea in ecology (Likens et al., 1977) that is particularly useful in human dominated ecosystems where there are goals to reduce nitrogen delivery to the atmosphere and receiving waters (Whitall and Paerl, 2001).

Wollheim et al. (2005) found 65–85% retention in urban watersheds in Massachusetts, with lower values in a wet year than a dry year. Their analysis highlighted the difficulty of closing the water budget in urban watersheds due to transfers into and out of basins. Kaushal et al. (2008a) found that retention in urban and suburban watersheds in Baltimore varied markedly with climate, declining from 85% in a dry year to 35% in a wet year. These studies indicate the need for long-term data in urban watersheds, as the hydrologic and biogeochemical drivers of nitrogen retention are complex, variable, and respond dynamically to climate variation.

Carbon and energetic budgets are closely related. Clearly, urban areas are heterotrophic ecosystems that depend on productivity from elsewhere (Collins et al., 2000). Cities in industrial countries may use between 100,000 and 300,000 Kcal m^{-2} yr^{-1}, whereas solar powered ecosystems typically expend between 1000 and 10,000 Kcal m^{-2} yr^{-1} (Odum, 1997). The massive energy conversion rates of cities are driven by fossil fuel subsidies, and contribute to the urban heat island discussed earlier (Section 3.1).

In spite of the massive respiration or combustion of carbon based fuels in urban biota, industry, transport, and households, there is the potential for carbon sequestration in cities. For example, in Chicago, trees have been estimated to sequester 5575 metric tons of air pollution, and 315,800 metric tons of C per year for a per ha rate of 17 metric tons (McPherson et al., 1997). In the Mediterranean climate of Oakland, California, trees sequester 11 metric tons of C per year (Nowak, 1993). Natural forests on average sequester 55 metric tons per year (Zipperer et al., 1997). The capacity of trees to filter particulates from urban air is based on leaf size and surface roughness (Agrawal, 1998).

Carbon flux has received increasing attention in urban ecological research with the widespread acknowledgement of the reality of global climate change. Coutts et al. (2007) examined CO_2 flux in two residential communities at daily, seasonal, and yearly scales in Melbourne, Australia. The neighborhoods differed in the amount of vegetation they supported. During the summer months, the commute signal was visible in morning and afternoon in both neighborhoods. The neighborhood with more trees also had more cars during commute, so the morning peak was higher than in the other neighborhood. In the afternoon, the peak in the neighborhood with more trees and more cars was lower than the other neighborhood because rates of photosynthesis were high.

An example of disentangling aspects of the carbon budget along an urban–rural gradient comes from Salt Lake City, Utah (Pataki et al., 2007). Over time, the highest values of CO_2 occurred in winter under atmospheric inversions. Using stable isotopes of C and O, it was discovered that greater than 50% of the CO_2 was contributed by combustion of natural gas in the wintertime, especially near the center of the city. During the growing season, more than 60% of the CO_2 originated from biological respiration, especially in rural sites. The O_2 isotope analysis further showed an enrichment attributable to irrigation in the urban sites. Leaf respiration contributed differentially to CO_2 in a residential neighborhood, but showed reduced seasonal variation in the less vegetated city center. This study shows considerable temporal and spatial variation in the fluxes that make up urban material budgets. Similar heterogeneities in material fluxes in metropolitan areas are shown by Golubiewski (2006) and Coutts et al. (2007). According to Pataki et al. (2006), urban carbon budgets are a maturing topic, with wide use of models combining land use, emissions from vehicular traffic, and the gains and losses from vegetation and soils. Urban carbon model validation has relied on both fine scale metabolic studies and atmospheric monitoring, and has been related to both biophysical and socio-economic drivers.

7.2. Urban footprints and their critiques

The budgetary approach, such as employed for nitrogen or carbon (Section 7.1) can be spatially expanded by considering the relationship of the energy and materials used in an urban area to areas beyond its borders. The ecological footprint of an urban area indexes the amount of productive land and sea required to support the material and energetic budgets required by a metropolis, and to process the wastes it generates (Rees, 1996). The amount needed is not spatially explicit meaning that it is the amount of land or water needed regardless of where it is located. For example, the city of Vancouver, Canada, requires 180 times more land to generate and process materials than the city actually occupies. The concept is highly metaphorical, because the actual networks from which any particular city draws resources, and the areas affected by its waste may extend around the globe (Opschoor, 2000; Luck et al., 2001). Limitations of the concept as an environmental accounting tool are that it denies the value of trade, the ecological characteristics of immediate hinterlands, and the economies of scale in urban areas that may reduce negative ecological impacts. Highly productive monocultures are given a higher capacity score, yet suffer from reduced biodiversity. An analysis of the growth of Chicago (Cronon, 1991) documented networks of resource acquisition that extended throughout the western regions of the late 19th century US. The metropolis in the post-industrial, information age in nations enjoying a high fossil fuel subsidy, sends different sorts of tentacles into the hinterland than did the industrially and agriculturally anchored Chicago in the late 19th and early 20th centuries (Bradley, 1995). Telecommuting, materially and energetically subsidized recreation, and the alteration of land values for “urban” uses in the countryside represent a footprint based on urban capital.

7.3. Pollution and nutrient dynamics

The high levels of pollution in urban areas have the potential to drastically affect nutrient cycling. How pollutants affect urban biogeochemical fluxes is uncertain, but preliminary results suggest that the effects depend on various urban environmental factors (Carreiro et al., 2009). For example, Inman and Parker (1978) found slower leaf litter decay rates in urban stands that were highly contaminated with Cu (75.7 mg kg^{-1}) and Pb (399.9 mg kg^{-1}) compared to unpolluted rural stands. Apparently, the heavy metal contamination negatively affected the activity of decomposers in these urban stands. By contrast, where heavy metal contamination of soil was moderate to low relative to other atmospherically deposited pollutants such as N, rates of decomposition and soil N-transformation increased in forest patches near or within major metropolitan areas in southern California (Fenn and Dunn, 1989; Fenn, 1991), Ohio (Kuperman, 1999) and southeastern New York (Carreiro et al., 2009). An experimental demonstration of how pollutants can condition the response of urban plants appears in the work of Gregg et al. (2003). Using uniform clones of hybrid poplar trees, growth was seen to decrease downwind from New York City. Ozone was the predominant control on growth of these clones in common garden experiments along the urban–rural gradient rather than increased N deposition or temperature toward the city center. Armentano and Loucks (1983) also documented distant effects of urban ozone (see Section 3 for other climate effects).

7.4. Urban convergence: lawns and soil carbon

A central principle in urban ecological theory implies that anthropogenic drivers will dominate natural drivers in the control of ecosystem response variables (Pickett et al., 2008). The implication is that urban ecosystem and soil responses to urbanization should converge on regional and global scales relative to the native systems they replace. This idea is labeled the *Urban Ecosystem Convergence hypothesis* (Pouyat et al., 2003). Converting forest, grasslands, and deserts to urban and suburban land uses entails a complex array of ecosystem alterations. The effects of these changes on soil organic carbon (SOC) are an often missing component in local, regional, and global estimates of SOC stocks (Kaye et al., 2005; Pouyat et al., 2006). Convergence in soil carbon stocks or other response variables will occur as long as the anthropogenic drivers and effects resulting from management, use, and environmental change, dominate native controlling factors, such as topography and parent material.

An important characteristic of urban land-use conversion with respect to the carbon cycle is the introduction of turf grass cover (Kaye et al., 2005; Milesi et al., 2005; Golubiewski, 2006). With approximately 8% of the nation’s land base in urban use (CIESIN, 2004) the total estimated areal amount of turf grass for the conterminous United States is 163,800 $km^2 \pm 35,850$ km^2, which is estimated to exceed by three times the area of other irrigated crops (Milesi et al., 2005). Residential and institutional managers apply about 16 million kg of pesticides each year (Aspelin, 1997) as well as fertilizers at rates similar to or exceeding those of cropland systems, e.g., up to 200 kg ha^{-1} yr^{-1} (Law et al., 2004).

Due to the efforts of residential and other land managers, the introduction of turf grass and corresponding management should result in a significant accumulation of SOC in most urban landscapes. Qian et al. (2003) using the CENTURY model showed that N fertilization coupled with grass clipping replacement increased SOC accumulations by up to 59% compared to sites that were not fertilized and from which clippings were removed. Likewise, Golubiewski (2006) measured SOC stocks in 13 residential yards of different ages built on the semiarid shortgrass steppe of Colorado and found that SOC recovered from the initial disturbance caused by development after 20 years, and exceeded the semiarid prairie soils shortly thereafter. In addition, the effects of lawn management on SOC accumulation exceeded the effects of other soil forming factors, such as elevation and soil texture.

The amount of effort to manage turf grass systems, e.g., intensity of irrigation or nutrient applications, should also have an effect on SOC accumulation. Moreover, the effort should reflect the natural constraints on the soil-turf grass system (Pouyat et al., 2008a). For example, irrigation rates for turf grass growing in dry land areas must be higher than for turf grass in more temperate climates. Indeed, in the absence of irrigation and fertilization, most species of turf grass would not be able to grow and compete with native vegetation in most of the conterminous US (Milesi et al., 2005). Therefore, as the constraints on the growth of turf grass increase, the management effort will also have to increase so that turf grass species can compete favorably with other plants (Pouyat et al., 2008a). Therefore, compensatory inputs of water and nutrients required to maintain turf grass cover should result in a higher accumulation of SOC than in the original dry land system (e.g., Golubiewski, 2006; Jenerette et al., 2006). Whereas, in more temperate regions of the US where turf grasses may grow without supplements, SOC in residential areas should be equivalent to or lower than the native soil, which will most likely be a forest soil. Thus, SOC should converge from a wide range of starting conditions in urban areas (Pouyat et al., 2008b).

To address the SOC convergence hypothesis, a continental analysis of available SOC data for urban and native soils by Pouyat et al. (2006) showed that remnant patches of native vegetation accounted for almost 10% of the urbanized area in the conterminous US, and, depending on the SOC density (kg C m^2-1 m depth) of the native soils, could account for up to 34% of the stock of SOC of a city. Moreover, when soils beneath impervious surfaces were excluded from the analysis, the estimated SOC densities rose substantially for the urban areas, indicating the potential for soils in pervious areas of urban landscapes to sequester large amounts of SOC. The authors also compared the pre-agricultural, agricultural, and post-urban estimates of SOC stocks in six cities from which data were available and found the potential for large decreases in SOC pools post-urban development for cities located in the northeastern U.S. where native soils have inherently large SOC densities, whereas in drier climates, cities tended to have slightly more SOC after urban development (Pouyat et al., 2006). In a follow up to the continental analysis, Pouyat et al. (2008b) found a similar relationship between residential and native soils in the Baltimore and Denver metropolitan areas.

7.5. *Invasive species and biogeochemistry*

Invasive species can play a disproportionate role in controlling C and N cycles in terrestrial ecosystems (Bohlen et al., 2004; Ehrenfeld, 2003). Therefore, the relationship between invasive species abundances and urban land-use change has important implications for biogeochemical cycling of C and N (Pouyat et al., 2008a). For example, in the northeastern and mid-Atlantic United States where native earthworm species are rare or absent, urban areas are important foci of non-native earthworm introductions, especially Asian species from the genus *Amynthas*, which are expanding toward outlying forested areas (Steinberg et al., 1997; Szlavecz et al., 2006). These invasions into forests have resulted in highly altered C and N cycling processes (Bohlen et al., 2004; Carreiro et al., 2009). Invasives of various life forms in urban areas have altered C and N cycles, including the shrub *Berberis thunbergii*, the tree *Rhamnus cathartica*, and the grass *Microstegium vimineum* (Ehrenfeld et al., 2001; Heneghan et al., 2002; Kourtev et al., 2002).

8. Human actors in urban ecosystems

In introducing urban ecosystems, we noted that humans might be included and discussed among the other urban organisms. However, we have chosen to discuss humans after introducing the other state factors for two reasons: Humans are the dominant organisms in urban systems, and their ecology is especially complex and highly differentiated, with many layers of social structures and dynamics. Thus, social scientists were among the first scholars to take an ecological approach to urban areas. The social scientists of the Chicago School, which was active in the early 20th century, used analogies with ecological theories of succession and competition (Light, 2009). Perhaps ironically, the flaws that emerged from the application of classical equilibrium theory from ecology to urban sociology produced frameworks that were severely criticized in the social sciences (Gottdiener and Hutchison, 2000). New social theories which account for spatial, social and temporal differentiation provide a sound basis for an ecological approach to urban systems.

8.1. *Social ecology and social differentiation*

The study of social structures and how those structures come to exist is needed for the integrated study of urban ecology. It is increasingly difficult to determine where biological ecology ends and social ecology begins (Golley, 1993). Indeed, the distinction between the two has diminished through the convergence of related concepts, theories, and methods in the biological, behavioral, and social sciences. Social ecology is a life science focusing on the ecological study of various social species such as ants, wolves, or orangutans. We may also study humans as an individual social species or comparatively with the ecology of other social species. The subject matter of social ecology, like biological ecology, is stochastic, historic, and hierarchical (Grove and Burch, 1997). In other words, living systems are not deterministic; they exhibit historical contingencies that cannot be predicted from physical laws alone (Botkin, 1990; Pickett et al., 1994).

8.2. *Assumptions of social theory and links to ecology*

Human ecology must reconcile social and biological facts to understand the behavior of people over time (Machlis et al., 1997). Such a biosocial approach to human ecological systems (Burch, 1988; Field and Burch, 1988; Machlis et al., 1997) stands in contrast to a more traditional geographic or social approach (see Hawley, 1950; Catton, 1994). This is not to say that social sciences such as psychology, geography, anthropology, sociology, economics, and political science are unimportant to social ecology. They are, since the most fundamental fact that distinguishes humans and their evolutionary history from other species—both social and non-social—is that human social development has enabled the species to escape local ecosystem limitations. Local ecosystems no longer regulate human population size, structure, or genetic diversity (Diamond, 1997). Nowhere is this more apparent than in urban ecosystems.

8.3. *Social differentiation: causes and ecological effects*

Theoretical development requires a closer look at the varieties of social differentiation that can exist within urban ecosystems. All social species are characterized by patterns and processes of social differentiation (van den Berghe, 1975). In the case of humans, social differentiation or social morphology has been a central focus of sociology since its inception (Grusky, 1994). In particular, social scientists have used social identity based on such features as age, gender, class, caste and clan, and the social hierarchies based on these features, to study how and why human societies become differentiated (Burch and DeLuca, 1984; Machlis et al., 1997).

Social differentiation is important for human ecological systems because it affects the allocation of critical resources, including natural, socio-economic, and cultural resources. In essence, social differentiation determines "who gets what, when, how and why" (Lenski, 1966; Parker and Burch, 1992). This allocation of critical resources is rarely equitable, but instead results in rank hierarchies. Unequal access to and control over critical resources is a consistent fact within and between households, communities, regions, nations and societies (Machlis et al., 1997). Environmental justice scholarship has demonstrated that disadvantaged groups, especially racial and ethnic minorities, are disproportionately burdened with environmental disamenities and enjoy fewer amenities compared to the privileged majority (Mohai and Saha, 2007). These inequitable outcomes are typically the result of unjust procedures that burden racial and ethnic minorities with the most polluted and hazardous environments close to where they live (Bolin et al., 2005).

Five types of sociocultural hierarchies are critical to patterns and processes of human ecological systems: wealth, power, status, knowledge and territory (Burch and DeLuca, 1984). Wealth consists of access to and control over materials in the form of natural resources, capital or credit. Power is the ability to alter others' behavior through explicit or implicit coercion (Wrong, 1988). The powerful have access to resources that are denied the powerless. One example is politicians who make land-use decisions or provide services for specific constituents at the expense of others. Status is access to honor and prestige and the relative position of an individual or group in an informal hierarchy of social worth (Lenski, 1966). Status is distributed unequally, even within small communities, and high-status individuals may not necessarily have access to either wealth or power. Knowledge is access to or control over specialized types of information, such as technical, scientific, and religious. Not everyone within a social system has equal access to different types of information. Knowledge often provides advantages in terms of access to and control over the critical resources and services of social institutions. Finally, territory is access to and control over critical resources through formal and informal property rights (Burch et al., 1972; Bromley, 1991).

8.4. *Households as a key node in social differentiation*

It has become clear that households and their behavior are important to biophysical characteristics. This is in contrast to the simplistic view that population density is an adequate predictor of environmental relationships. Liu et al. (2003) found that the number of households increases much faster than the total population and this rapid increase has important implications for biodiversity and natural resource consumption. Oldfield et al. (2003) noted a relationship between household participation in outdoor recreation and household land management practices. While the shift in emphasis from total population size to households as a unit of analysis is significant and positive, a multi-dimensional characterization of households will enable a more complete understanding of the motivations, pathways, impacts, and responses of households to ecological change.

Extensive social science research on household behavior indicates that households are in fact multi-dimensional. For instance, households have been characterized using social class and lifestyle (Blumin, 1989; Higley, 1995), reference groups (Merton and Kitt, 1950; Shibutani, 1955; Singer, 1981), and consumer activity (Veblen, 1981 [1899]; Horowitz, 1985; Schor and Holt, 2000; Matt, 2003). These characteristics may explain variations among households in the types of employment they seek, what they choose to buy, where they choose to live, how they organize through participation in formal and informal associations, and how they spend their leisure time.

Burch and DeLuca (1984) describe how household characteristics such as housing and settlement preferences, household size and lifestage, cultural traditions, access to power and knowledge, and group identity and status can influence social and biophysical structures and functions. These interactions can be described and examined in a human ecosystem context (e.g., Machlis et al., 1997; Redman et al., 2004). As human ecosystem research is applied to urban areas, there is an increasing need to answer the question of whether the "usual suspects" among social variables – population density, income and education, and ethnicity (Whitney and Adams, 1980; Palmer, 1984; Grove and Burch, 1997; Dow, 2000; Vogt et al., 2002; Hope et al., 2003) – are adequate explanatory variables (Grove et al., 2005).

8.5. *Group identity and the ecology of prestige*

One way to understand the social drivers of household behavior and land management is to consider the social ecology of group identity and social status associated with different lifestyle groups and its significance to the ecological structure of urban environments (Grove and Burch, 1997; Grimm et al., 2000, Grove et al., 2006a,b). Housing styles, yard characteristics, tree and shrub plantings, green grass, and type of lawnmower can be considered social-ecological symbols, reflecting the type of neighborhood in which people live. These social-ecological symbols can be fruitfully interpreted as the outward manifestation of each resident's placement in a given social hierarchy of group identity and social status. The desire for a thick green lawn might not be some irrational whim. Rather, what drives a person to fertilize, pesticide, and herbicide might be exceptionally rational: it might uphold the prestige of the community and outwardly express each household's membership in a given territorial group. A person's chosen "lifestyle status symbols" are a manifestation of this belonging. In short, what people wear, drive, and inhabit are best seen as choices made within a specific social context rather than idiosyncratic market-based consumer decisions. There is not a single "mass market," but a diverse array of clustered, lifestyle niches, each representing a nuanced set of different group identities and types of social status (Jenkins, 1994; Weiss, 2000; Holbrook, 2001).

An example of how social differentiation can be used to understand the structure and function of an urban area explores how vegetation structure and cover relates to social characteristics across Baltimore. There are a variety of theories that can explain the linkage between vegetation structure and social phenomena. We focus on 1) property regimes, 2) lifestyle characteristics, and 3) classic social variables of stratification based on income and education. To test the theories, we distinguished among vegetation in areas hypothesized to be differentially linked to these three social theories: riparian areas, private lands, and public-rights-of-way. Using a multi-model inferential approach (Grove et al., 2006b; Troy et al., 2007), we found that variation of vegetation cover in riparian areas was not explained by any of the three theories, while lifestyle behavior was the best predictor of vegetation cover on private lands. Surprisingly, lifestyle behavior was also the best predictor of vegetation cover in public-rights-of-way. The inclusion of a quadratic term for housing age significantly improved the models. These results question the exclusive use of income and education as standard variables to explain variations in vegetation cover in urban ecological systems, and indicate the applicability of several theoretical models, depending on location within the urban mosaic. These results also suggest that management of urban vegetation can be improved by accounting for environmental marketing differentiation that underlies household motivations for and participation in local land management (Fig. 5).

8.6. Space, scale, and social differentiation

Social differentiation of human ecological systems has a spatial dimension characterized by territoriality and heterogeneity (Morrill, 1974; Burch, 1988). As Burch (1988) notes, "Intimate and distant social relations, high and low social classes, favored and despised ethnic, occupational, and caste groupings all have assigned and clearly regulated measures as to when and where those relations should and should not occur". When ecosystem and landscape approaches are combined, the research changes from an ecosystem budgetary question of "who gets what, when, how and why?" to a question of "who gets what, when, how, why and *where*?" and, subsequently, "what are the reciprocal relationships between spatial patterns and sociocultural and biophysical patterns and processes of a given area" (Grove, 1997)? This development parallels evolution of the biological concept of ecosystems to include spatial processes, and provides a link between the two disciplines.

Various processes of social differentiation occur at different scales and have corresponding spatial patterns and biophysical effects (Grove and Hohmann, 1992). Based on social and ecological theory, examples include global and regional urban–rural hierarchies (Morrill, 1974), the distribution of land uses within urban areas (Guest, 1977), the stratification of communities within residential land uses (Logan and Molotch, 1987), and the social differentiation of ownerships and households within communities (Burch and Grove, 1993; Grove, 1995).

8.7. The power of parcels: new data and social approaches

The spatial structure of social differentiation requires a multiscalar strategy for sampling in order to integrate biophysical and social structures and processes in metropolitan ecosystems. We have discovered that a parcel-based sampling plan that combines both extensive and intensive sampling is appropriate to the complex structure (Cadenasso et al., 2006b) of metropolitan systems. The motivations for using parcels as a fundamental unit are several. First, parcels are a basic unit of decision-making associated with household and firm locational choices and behaviors. Second, parcels and their owners have social and ecological histories, and their geographies and attributes can be documented and described over time through a variety of sources (Boone et al., 2010). Third, parcels can be aggregated into other units of analysis, such as patches, neighborhoods, watersheds, and municipalities (Grove et al., 2006a). Disaggregation is also possible, for example, to investigate differences between biophysical and social features of front and back residential yards (Loucks, 1994).

The empirical ability to examine and integrate social and ecological characteristics at a parcel level and regional extent is relatively new. Until recently, only coarse resolution geospatial data had been available. For instance, regional vegetation cover data had typically been derived from 30 m resolution Landsat Thematic Mapper (TM) satellite imagery. Socio-economic analyses had normally been carried out at the US Census Tract level in which a single tract contains approximately 2500–8000 persons or a US Census Block Group, which contains between 200 and 400 households. Other geospatial data, particularly cadastral information, had more often than not been maintained by local governments in hardcopy format (Troy et al., 2007).

Recent advances in remote sensing and the widespread adoption of Geographic Information Systems (GIS) by federal, state, and local governments have increased the availability of high-resolution geospatial data. Vegetation characterization can be derived

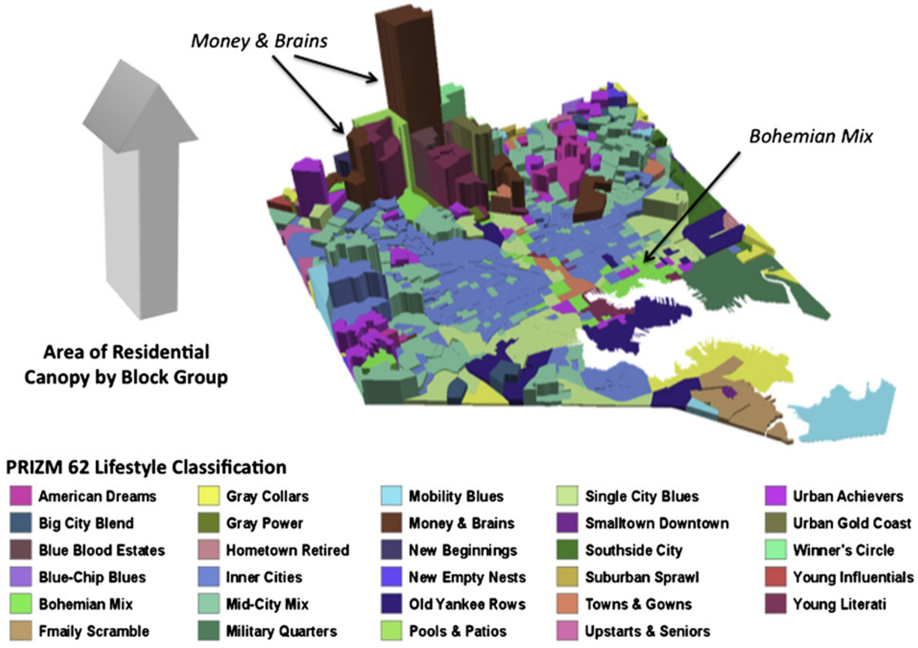

Fig. 5. An ecology of prestige and urban tree canopy (UTC). US Census Block Groups were classified using the Claritas PRIZM market classification system and percent canopy cover for residential areas in each Block Group (Grove et al., 2006a,b). While including population density and socio-economic status, the PRIZM 62 Lifestyle Classification also includes household lifestage and housing type. Although both PRIZM's "Money and Brains" and "Bohemian Mix" have similar levels of population density and socio-economic status, they are significantly different in terms of household lifestage and housing type. "Money and Brains" neighborhoods also have 47% more canopy cover in residential areas than "Bohemian Mix" neighborhoods (Troy et al., 2007).

from high-resolution imagery and combined with digital parcel geographies, which includes property boundaries for each parcel, and digital surface water data to distinguish among vegetation extent, structure, and productivity on private property and public-rights-of-way, including along streets. These parcel data also include parcel attributes such as building type, features, age, transacted value, quality, owner, and parcel land use and zoning (Zhou et al., 2008b).

These empirical advances provide a foundation for combining traditional social and ecological data. For instance, different types of surveys can be linked to these data by both a unique address and by latitude and longitude. In the case of telephone surveys, telephone lists can include address and spatial location, while field observations and interviews can record both address and Global Positioning System (GPS) location as surveys are conducted. All of these data can be linked to Census geographies, which provide the basis for including demographic and socio-economic data from the Census and marketing data that are available at the Block Group level. Some of these marketing data include residential land management behaviors such as household expenditures on lawn-care supplies and services (Zhou et al., 2008a, 2009). These data can be further combined with a variety of present and archival data that are address based, including real estate transactions, business directories, legal and health records, biographies and diaries, photographs, and neighborhood association minutes, for example.

The sampling strategy is completed by spatially extensive sampling. This provides the basis for stratifying more spatially intensive data sampling of social and ecological data, for example those using direct field measurements or in-depth interviews. In some cases, intensive sampling focuses on specific parcels (Lidman, 2008). In other cases, such as hydrological and atmospheric measurements, the location of sampling points can be based upon strategic combinations of parcels based upon characteristics such as population density, socio-economic status, lifestyle characteristics, zoning, land use, soils, or topography.

9. Frameworks and integration

There are several approaches to integrate the state factors that we have, for simplicity, treated as largely distinct to this point. Many research projects have attempted to bring mainstream ecology and crucial social sciences more closely together (Pickett et al., 1997b; Grimm et al., 2000; Alberti et al., 2003, Redman et al., 2004). When we began to contribute to this research agenda through the Baltimore Ecosystem Study in 1997, it was important to employ familiar concepts that each discipline could embrace. Hence, we began with the ecosystem as a tool to organize research and conceptualize an urban area as an interdisciplinary research topic (Cadenasso et al., 2006a).

9.1. The ecosystem

The ecosystem concept and its application to humans is important because of its utility for integrating the physical, biological, and social sciences. The ecosystem concept owes its origin to Tansley (1935), who noted that ecosystems can be of any size, as long as the concern was with the interaction of organisms and their environment in a specified area. Further, the boundaries of an ecosystem are drawn to answer a particular question. Thus, there is no set scale or way to bound an ecosystem. Rather, the choice of scale and boundary for defining any ecosystem depends upon the question asked and is the choice of the investigators. In addition, each investigator or team may place more or less emphasis on the chemical transformations and pools of materials drawn on or created by organisms; or on the flow, assimilation, and dissipation of biologically metabolizable energy; or on the role of individual species or groups of species on flows and stocks of energy and matter. The fact that there is so much choice in the scales and boundaries of ecosystems, and how to study and relate the processes within them, indicates the profound degree to which the ecosystem represents a research approach rather than a fixed scale or type of analysis (Allen and Hoekstra, 1992; Pickett and Cadenasso, 2002).

A particular manifestation and context of the ecosystem concept is the watershed. Hydrologists examine how the abiotic attributes of different patches within a watershed—such as temperature and physical characteristics including topography, soil properties, water table depth and antecedent soil moisture—contribute variable amounts of water and nutrients to streamflow, depending upon their spatial location in the watershed (Black, 1991). Hydrologists have summarized mosaics of the characteristics listed above using the Variable Source Area (VSA) approach. The VSA approach can be integrated with a delineation of patches based upon the biotic attributes of the watershed—such as vegetation structure and species composition (Bormann and Likens 1979)—and the social attributes of the watershed—such as indirect effects from land-use change, forest/vegetation management and direct effects from inputs of fertilizers, pesticides and toxins—to examine how the abiotic, biotic and social attributes of different patches within a watershed contribute variable amounts of water and nutrients to streamflow (Grove, 1996). This integrated approach builds on the VSA approach introduced by hydrologists to combine nested hierarchies of land use and land cover, socio-political structures, and hydrological heterogeneity.

9.2. The human ecosystem framework

An additional integrative framework emerged from the social sciences. Social scientists have focused on interactions between humans and their environments since the origin of their discipline. However, the explicit incorporation of the ecosystem concept within the social sciences dates to Duncan's (1961, 1964) articles "From Social System to Ecosystem" and "Social Organization and the Ecosystem". Recently, the social sciences have focused increasingly on the ecosystem concept because it has been used for natural resource policy and management (Rebele, 1994).

Although the ecosystem concept is flexible enough to account for humans and their institutions (Tansley, 1935; Rebele, 1994; Pickett and Grove, 2009), the application of an ecosystem approach to the study of human ecosystems requires an additional analytical framework. The analytical framework we use here (see Burch and DeLuca, 1984; Machlis et al., 1997; Pickett et al., 1997b) is not a theory in and of itself. Rather it can be cast as a causal hierarchy (Fig. 6) that identifies the specific kinds of variables and interactions to be included in urban ecological models. Machlis et al. (1997) note that some features of the framework are "orthodox to specific disciplines and not new". However, they also indicate that it also contains some less commonplace features such as myths as cultural resources, or justice as a critical institution. We adopt their view that the hierarchical framework is a coherent entity that is useful in structuring the study of human ecosystems.

There are several elements that are critical to the successful application of this framework. First, it is important to recognize that the primary drivers of human ecosystem dynamics are both biophysical and social. Second, there is no single, determining driver of anthropogenic ecosystems. Third, the relative significance of drivers may vary over time. Fourth, components of this framework need to be examined simultaneously in relation to each other (Machlis et al., 1997). Finally, researchers need to examine how dynamic biological and social allocation mechanisms such as

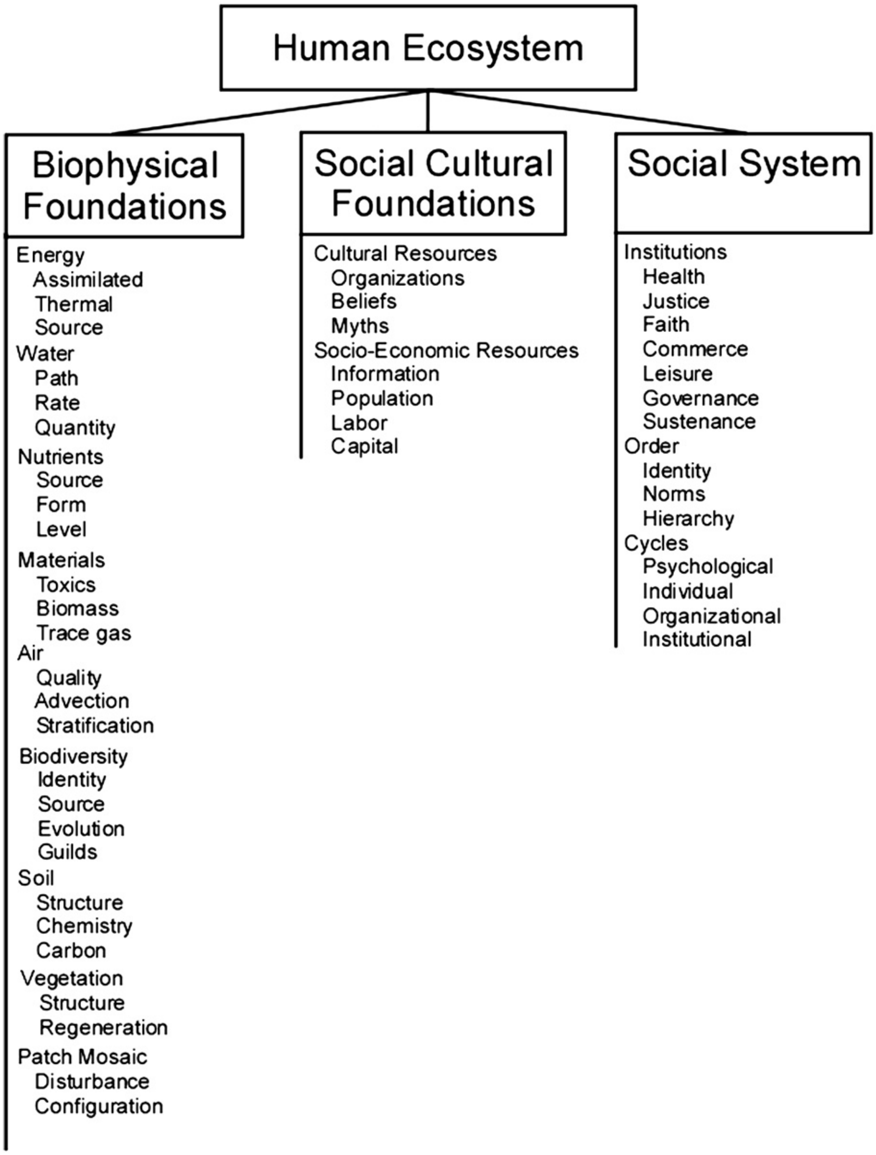

Fig. 6. Human ecosystem framework as a causal hierarchy. The framework illustrates the kinds of structures and the interactions that can exist in an integrated human-natural system. The features of the framework are divided into the biophysical foundations, the social and cultural foundations, and the social system that emerges from the social interactions and the interactions of humans and their institutions with the biophysical foundations. Modified, with additions to the biophysical component, from Machlis et al. (1997).

ecological constraints, economic exchange, authority, tradition and knowledge, affect the distribution of critical resources including energy, materials, nutrients, population, genetic and non-genetic information, labor, capital, organizations, beliefs and myths, within any human ecosystem (Parker and Burch, 1992).

The human ecosystem concept requires two things to be operationalized in urban areas. The first is the causal framework laid out above as the human ecosystem framework (Fig. 6). The second requirement is a template for how the causes and factors identified in the human ecosystem framework might actually interact to generate the complex structures and functions of urban areas. Of the many human ecosystem model templates currently available, we select one as an illustration (Fig. 7). All model templates for urban ecosystems recognize the reciprocal relationships of biological structures and processes, socio-economic structures and processes, slowly changing historical or evolutionary templates, and global or regional external drivers. Furthermore, they acknowledge the role of social differentiation and the perception by individuals or institutions as mediators of the interactions between biophysical and socio-economic patterns and processes. Feedbacks, often with time lags and indirect effects, are a part of these conceptual model templates of urban ecosystems.

In order to generate a specific model to translate any model template and causal framework into an operational model, it is crucial to take into account the spatial heterogeneity or complexity within urban systems (Cadenasso et al., 2006b). A new way to conceptualize the structural heterogeneity of urban systems builds upon the three dimensions of structure recognized in geography (Ridd, 1995): 1) built structures, 2) surfaces, and 3) vegetation. Cadenasso et al. (2007a) developed the HERCULES (High Ecological Resolution Classification for Urban Landscapes and Environmental Systems) land-cover classification to express the integrated nature of spatial heterogeneity that must be considered in translating a model template to an operational form. Their land-cover classification identifies patches based on the type and proportional cover of buildings present, the proportion of paved or bare surfaces, and

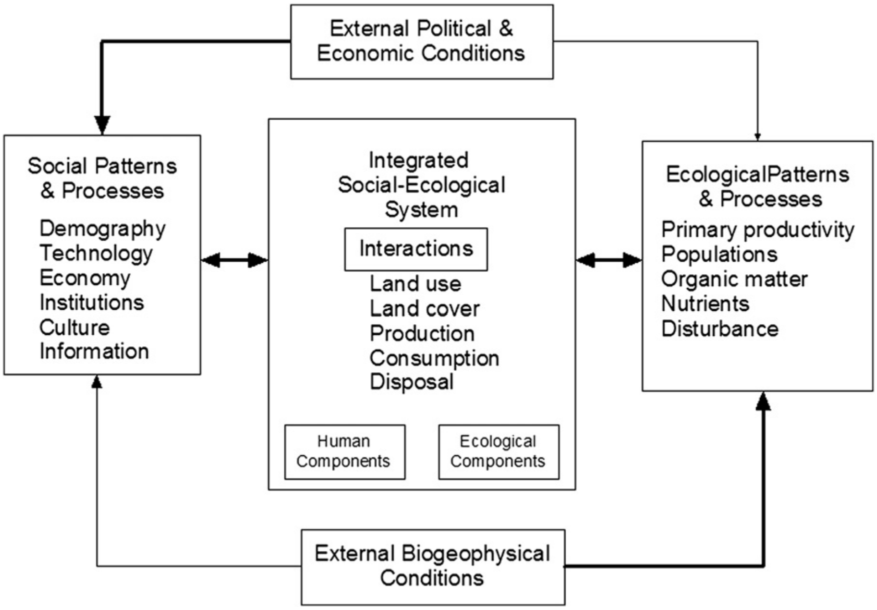

Fig. 7. A human ecosystem model template. The model specifies a focal concern with an integrated social-ecological system, which depends on both social patterns and processes and biophysical ecological patterns and processes. External social conditions and external biophysical conditions are capable of modifying the system through their effects on either the social or the ecological patterns and processes. The structural and interactive details from the human ecosystem causal framework (Fig. 7) can be used to suggest the content of a specific human ecosystem model. Modified from Redman et al. (2004).

the proportion of woody or herbaceous cover. It is important that the classification, unlike the traditional methods used in urban systems, does not assume that vegetation and anthropogenic structures are mutually exclusive. While the traditional land-use/land-cover models assumed a general, high level segregation of urban structure and vegetation, conceiving of urban covers as complex and integrated has value. Acknowledging that the kind and proportion of buildings, the kind and cover of surfaces, and the kinds and cover of vegetation can vary relative to one another portrays urban heterogeneity in a flexible and realistic way (Fig. 8). Indeed, the multi-dimensional land-cover classification has a better ability to represent ecosystem function compared to traditional land-use/land-cover classifications (Cadenasso et al., 2007a). We now turn to several modeling approaches that can be used to understand feedbacks between socio-economic components and biophysical components of human ecosystems.

9.3. Land-use science and modeling approaches

A human landscape approach may be understood as the study of the reciprocal relationships between patterns of spatial heterogeneity and sociocultural and biophysical processes. Further, when human ecosystem and landscape approaches are combined, human ecosystem types or patches are defined as homogeneous areas for a specified set of sociocultural and biophysical variables within a landscape. Analyses then focus on two issues: 1) the development and dynamics of spatial heterogeneity, and 2) the influences of spatial patterns on cycles and fluxes of critical ecosystem resources (e.g., energy, materials, nutrients, genetic and non-genetic information, population, labor, capital, organizations, beliefs or myths; Fig. 6). For instance, the development and dynamics of heterogeneity in a watershed spanning urban-to-rural conditions may influence and be influenced by sociocultural and biophysical processes. Patches within the watershed may function as either sources or sinks as well as regulate flows and cycles of critical resources between patches. The delineation and classification of these relatively homogeneous patches are based on a limited number of representative sociocultural and biophysical indicators (Burch and DeLuca, 1984; Parker and Burch, 1992) and traditionally studied as opaque boxes with fluxes and cycles of critical resources between areas (Zonneveld, 1989). The spatial linkages between the social and ecological differentiation of the watershed and their relationship to different types of allocation mechanisms at different scales are important for understanding the flows and cycles of critical resources within the watershed.

Land-use science is concerned with the study of land use and land cover, their changes over space and time, and the social, economic, cultural, political, decision, environmental and ecological processes that produce these patterns and changes (Aspinall, 2006). The objectives of this interdisciplinary research field, also

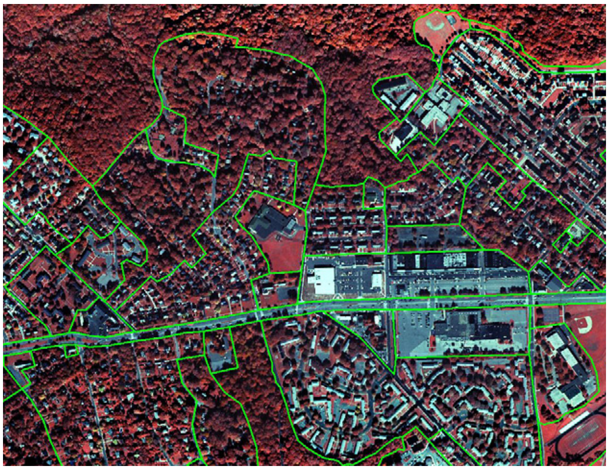

Fig. 8. Patches in an urban mosaic based on the High Ecological Resolution Classification for Urban Land and Environmental Systems (HERCULES; Cadenasso et al., 2007a) in western Baltimore City, MD. The classification is based on the nature and cover of vegetation, surfaces, and buildings, thus integrating biophysically and socially generated patch elements. Green lines on the image are boundaries between patches.

referred to as “land change science”, include observation and monitoring of land changes; understanding of these changes as a coupled human-environment system; spatially-explicit modeling of land change and assessment of land change outcomes (Fig. 9), e.g., including resilience and sustainability (Turner et al., 2007). Land-use and land-cover change are both the *result* of biophysical and human processes and an important factor in *determining* biophysical and human processes. For this reason, land-use science is necessarily an interdisciplinary science (Aspinall, 2006) and is undertaken by a variety of research communities, including remote sensing, political ecology, resource economics, institution governance, landscape ecology, biogeography, and integrated assessment, among others (Turner et al., 2007). Increasingly, land-use and land change systems are conceptualized as coupled human-natural systems modeled as complex systems characterized by critical two-way linkages within and between the human and biophysical subsystems (Parker et al., 2003; Turner et al., 2007). Therefore, land-use change research relies on the integration of both natural science and social science approaches that use spatially-explicit methods, including Geographic Information Systems (GIS), for data capture and model development (Aspinall, 2006).

Land use and land cover are the result of interactions among a variety of complex human and biophysical processes that operate across multiple spatial and temporal scales. For example, household location decisions are determined by a variety of economic, social, environmental, political and geographic factors that operate across a divergent range of spatial and temporal scales (Fig. 9). These processes are interdependent, exhibiting so-called cross-scale interactions that flow in bottom-up or top-down directions. Local interactions among individual agents, e.g., households, firms, and landowners, represent an important source of bottom-up interactions: a household’s location choice may be influenced not only by the “own” house or parcel features, but the neighboring house or land parcel attributes as well. In such cases a form of interaction arises that can generate systematic patterns of household sorting or land-use patterns at a more aggregate scale, such as segregation and sprawl. Alternatively, metropolitan-wide urban or natural amenities may induce migration of similar types of households into a region, which can have a top-down influence on the demand for local policies and public services at a neighborhood scale.

A variety of theoretical, empirical and simulation models have been developed to study the various patterns and processes associated with land-use and land-cover change (e.g., see Agarwal et al., 2002; Anas et al., 1998; Irwin and Geoghegan, 2001; Parker et al., 2003 and Verberg et al., 2004 for reviews). Theoretical models grounded in economic, geographic, political, sociological or anthropological theories of land use or location behavior most often focus on a particular behavioral dimension and explain land-use/land-cover outcomes as a result. For example, the basic urban bid-rent model posits that transportation costs to a central employment district bid up land rents in areas close to the employment center, which results in gradients of declining land rent and urban density. Alternatively, household sorting models focus on how heterogeneous households that differ, for example by income, race, ethnicity or other distinguishing features, sort themselves into different neighborhoods according to their ability and willingness to pay for neighborhood public goods. This in turn determines the important characteristics of the neighborhood, including the quality and mix of public goods within the neighborhood.

Theoretical models provide the conceptual framework for explaining the human or biophysical processes that underlie land-use and land-cover change. These frameworks are used to motivate and specify spatially explicit empirical land-use change models, which seek to identify the statistically significant associations between land-use change and spatially heterogeneous variables. A variety of estimation approaches of spatially explicit, micro-scale models of land-use change are possible. Spatially-explicit models at the parcel level include discrete choice models of rural-to-urban land-use change (e.g., Bockstael, 1996) and duration models that estimate the timing and location of land-use change by incorporating time-varying explanatory variables (e.g., Irwin and Bockstael, 2002). These models can then be used to generate predictions regarding future likelihoods of spatial land-use change under baseline and alternative policy scenarios. For example, Irwin et al. (2003) examine the predicted influence of growth management

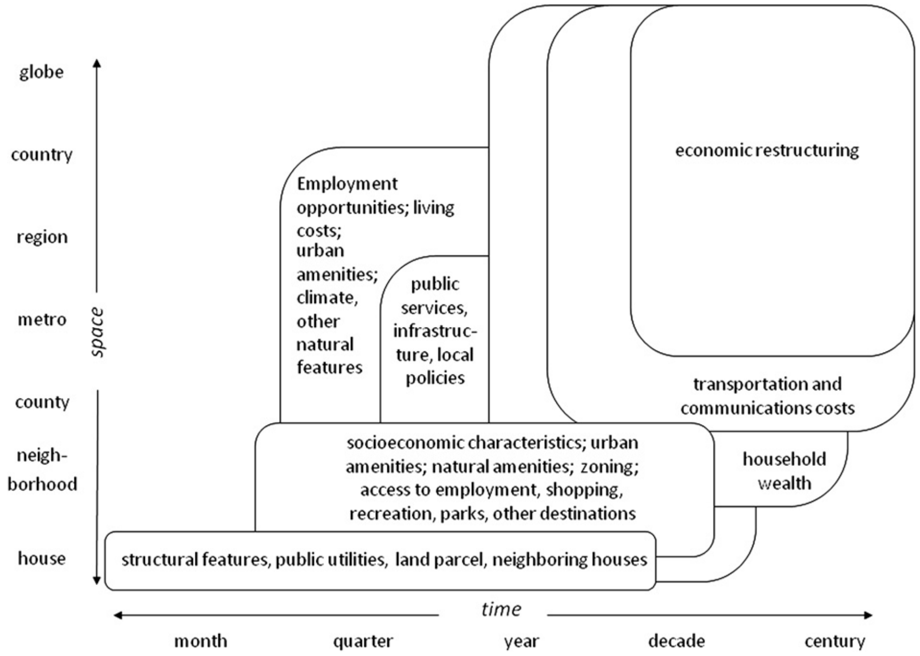

Fig. 9. Temporal and spatial scales of factors influencing household location decisions.

and agricultural preservation policies on residential sprawl patterns in an exurban region of central Maryland. Hedonic models of residential property values provide a related approach to explicitly modeling the factors influencing land values, which are often the prime determinants of land-use change (e.g., Geoghegan et al., 1997). Such spatially explicit approaches to modeling land-use change provide a useful framework for incorporating the biophysical spatial processes that interact with land-use/land-cover change (Costanza et al., 1990). For example, Lohse et al. (2008) use spatially-explicit parcel-level data to examine the influence of land use on salmon spawning substrate quality in tributaries of the Russian River in California and develop a land-use change model to forecast the probability of losses in high-quality spawning habitat.

Because of the complexity of the many spatial processes that underlie land-use/land-cover change and the inevitable limitations of available data in terms of measuring these processes, identifying causal effects of land-use change is extremely challenging. Techniques that employ spatial econometrics, quasi-experimental designs or instrumental variables provide a potential means for drawing causal inferences and can be extremely useful in isolating the effects of a spatially varying policy or a heterogeneous landscape feature on land-use change (e.g., Lynch and Liu, 2007; Baum-Snow, 2007).

A key challenge in modeling the processes that underlie land-use change is modeling cross-scale interactions that link individual-level processes with larger-scale outcomes and processes. These interactions are dynamic, usually non-linear and often path dependent, implying that future land-use patterns are determined by past and present outcomes. Statistical modeling alone cannot fully account for dynamic feedbacks and thus simulation methods are necessary for a comprehensive approach to modeling land-use change over time and space (Irwin et al., 2008). Cellular automata (CA) is one approach to simulating land-use dynamics. These models consist of spatially and temporally explicit fixed cells whose states are discrete and evolve according to simple transition rules. The rules determine how the cells within a local neighborhood, or at varying spatiotemporal scales, influence the state of each cell at a particular point in time and how such local interactions lead to global patterns of land use. Because these models are two-dimensional, multiple sources of heterogeneity can be incorporated, e.g., based on remotely-sensed data on land use and other spatial features (Torrens and O'Sullivan, 2001). Nonetheless, CAs are limited in their ability to model the underlying human behavioral and biophysical processes that may flow across the landscape and generate land-use change (Parker et al., 2003; Benenson and Torrens, 2004).

The integration of CAs with spatially-explicit agent-based system (ABS) models provides an effective means of modeling human and biophysical processes that can then be used to simulate land-use dynamics (Parker et al., 2003; Benenson and Torrens, 2004). Agents are treated as autonomous, interacting objects within a common environment that is defined by the CA. Agent decisions may be influenced by each other as well as by environmental conditions, which in turn generate dynamic feedbacks that bind agents and environment together in a co-evolving system. These models provide a means of linking micro-scale human or biophysical spatial processes to the emergence of land-use/land-cover patterns at more aggregate scales. For example, Brown et al. (2008) show that initial environmental heterogeneity draws residents to an exurban region, which in turn alters the spatial distribution of environmental amenities and generates feedbacks at higher spatial scales as commercial and retail services follow once a critical mass of residents is reached. The strategy of modeling land change raises the question of interaction between the practices of urban design and planning and the ecological aspects of urban complexes. Goals, intentions, accidents, and indirect effects are all imbedded in this question, which we will explore in the next section.

10. From sanitary city to sustainable city: the evolution of design and application of contemporary ecology

The state factors have been both intentionally manipulated and indirectly affected by humans in their pursuit of urbanization (Pouyat and Effland, 1999). In this section, we explore some of the main themes in urban design and planning that impact the state factors of the urban ecosystem. Urban ecology and urban design have developed in parallel, and have only been closely and regularly linked over the last two decades or so. This section follows the parallels, and indicates the power of integrating design and ecology.

The move from understanding ecology *in* the city to the ecology *of* the city is a shift toward a more integrated framework for urban ecology studies. This shift toward an integrated framework explored earlier (Section 1.1), parallels and compliments the development in urban design from planning *of* the city to planning *in* the city. The design of cities clearly reveals social attitudes toward ecology, integration and heterogeneity.

The Commissioners Plan of New York City (1807–1811) is a case in point. New York State appointed three commissioners who decided that a grid of 155 streets and 12 avenues creating 1694 blocks, was the most efficient way to plan the city for commercial development. The commissioners justified the lack of parks, gardens or open space by the presence of the two estuarial fingers of the East and Hudson Rivers which embrace the island of Manhattan, unlike the much smaller streams which pass through London or Paris. The scores of springs and streams on the island itself were of no value to the plan.

The blocks created by the Commissioner's grid are elongated rectangles which vary from 200 ft (60.7 m) wide, to between 600 (182.9 m) and 800 ft (293.8 m) long. This arrangement allowed for an efficient subdivision of each block into row houses along the streets measuring 20 ft wide and 50 ft deep – a dimension set by the structural limits of wood joists spanning parallel masonry bearing walls between properties. The plots along the Avenues were better suited for larger buildings and commercial activities. The blocks were oriented north–west, south–east, ensuring that sunlight enters rooms in both the front and back of all the buildings. When built out, the block of row houses formed a courtyard space 100 ft (30.4 m) wide by between 500 (152.4 m) and 700 ft (213.4 m) long, and between 30 and 40 ft (9.1–12.2 m) tall divided into between 40 and 60 private gardens.

The Commissioners Plan was both boldly visionary, and hopelessly inadequate, as it could anticipate neither the skyscraper city of today, nor the public health implications of the population that filled even half of the grid by the middle of the 19th century. Built between 1837 and 1842, the Croton waterworks brought fresh drinking water from 41 miles (66 km) north of the city before finally erupting in a decorative fountain in front of the old City Hall. At the same time, reformers such as Henry Jackson Downing called for a great park or greensward for the city and in 1853 the state authorized the City of New York to acquire 700 acres of blocks in the center of the city. Frederick Law Olmsted and Calvert Vaux won the commission to design the new Central Park around a reservoir in 1857. The design of these new public spaces – tied to new green and blue infrastructure – demonstrates a shift from planning *of* the city as an ideal ordering system for human economic activity to planning *in* the city which needed to balance human activities with natural resources. We will identify that shift as moving from Master Planning to Urban Ecosystem Design, and a shift from the two-dimensional abstraction of the layout of streets in blocks to designs which link environmental resources to human wellbeing and the public life of the city.

A century after the construction of Central Park, McHarg's (1969) *Design with Nature* and Bacon's (1969) *Design of Cities* were published at a time of a resurgence of Master Planning. In New York, Robert Moses reigned as a supreme power broker and master planner (Caro, 1974), creating a huge new network of highways, bridges, parks for active recreation and large swaths of public housing. Despite its title, McHarg's text and his work with Wallace, Roberts, Todd, was a treatise on a more ecologically sensitive master planning, which emphasized the value of incorporating knowledge of natural features when developing a regional plan. McHarg advocated mapping the city region on layers which separated various natural resources and risks, buffer zones and preservation areas. His technique, now aided by Geographical Information Systems and new visualization technologies (Grove, 1997; McGrath, 2008), became the normal practice of the planning and landscaping of America's exurban sprawl cities. Bacon's (1969) text justified his own massive urban renewal projects in downtown Philadelphia as Director of City Planning in relation to historical cities designed by emperors (e.g., Beijing and Paris), kings (e.g., London), and popes (Rome).

Social actors barely appear in McHarg's (1969) work, and are forcibly dislocated in the urban renewal planning surgery of Moses and Bacon. However, Jacobs' (1961) close observation of Harlem and her own brownstone neighborhood in New York's Greenwich Village at that time, demonstrated the rich social network intimately tied to the fine grain 19th century city planned by the Commissioners. Jacobs' (1961) book *The Death and Life of Great American Cities* is the primary text for urban design and planning *in* rather than *of* the city, and provides concrete evidence for the basis of an integrated framework in social ecology, social integration, spatial heterogeneity and human ecosystems. Jacobs was an activist as well as a writer, and was able to spark community organization against some of Moses' most audacious plans such as the Lower Manhattan Expressway which would have cut across the intricate neighborhoods of Jacobs' New York.

This sociocultural shift pushed back against the authority of government alone to shape the city, and introduced a wider array of urban actors to decision-making. The 1961 Master Plan for New York was never approved, but the city established community boards and a review process for both public and private development. Jonathan Barnett helped establish the Urban Design Group at New York's Department of City Planning in 1964. This new professional and academic discipline was established just a decade earlier at Harvard's Graduate School of Design. The pragmatic approach of the Urban Design Group abandoned master planning and moved to a system of urban design modeling and zoning incentives which included three dimensional bulk and open space studies that sought to weigh private gain against public good.

The combination of the two post-World War II forces of master planning: 1) exurban sprawl as a scenic landscape (McHarg, 1969) and 2) urban renewal (Bacon, 1969; Caro, 1974) has created considerable challenges and opportunities for urban ecology and design for the 21st century. The packed blocks of the Commissioners' grid in New York became considerably emptied by these twin forces of planning. However, starting in the 1970s, many urban vacant lots in New York were leased by community gardeners for $1 per year in the Green Thumb Program. Landscape Architect Spirn (1998) began to look at the relationship between old stream beds and empty lots in Philadelphia, and began to see an embedded pattern of relationship between historical legacies of natural processes which effect economy, health, and human community in the present. For instance, she showed how the forgotten environmental template of drainage networks continued to affect infrastructure and the social structure of a west Philadelphia neighborhood.

The legacies of Olmsted, McHarg, Moses, and Jacobs, together with Sprin's rediscovery of embedded socio-natural processes in the inner city, all enter into the contemporary urban ecosystem design work as exemplified by the Baltimore Ecosystem Study. Three neighborhoods in metropolitan Baltimore are being investigated in order to understand ecosystem services correlated with neighborhood design preferences. In the outer suburbs, BES has discovered that the planning principles, such as riparian buffers, advocated by McHarg do not necessarily create a healthy watershed (Section 3.5), as subdivisions with well functioning septic systems, 4 acre lots and generous stream buffers continue to export excess nitrogen (Groffman and Likens, 1994). While the streams run clear, they also run fast, and continue to incise the land. New designs would instead retain and spread the water around swampy and mucky wetlands rather than feed them to streams as if they were merely storm drains. In the older suburbs, urban parks do a better job of nitrogen retention than exurban riparian buffers, but dense neighborhoods of small houses and yards could be redesigned to manage rain water much better. Finally, in the inner city neighborhoods of row houses and vacant lots, both storm and sanitary sewer systems leak, and community greening is only locally considered. Infrastructure repair together with networks of open space greening projects tied to water retention gardens are designed with the whole watershed function in mind. Surveys with residents from these three neighborhoods will demonstrate the degree people are willing to institute changes in the design and management of urban neighborhoods.

Urban design and planning *in* the city, therefore, is much better positioned to be tied with the ecology *of* the city. The integrated framework for urban and social ecology, social integration, spatial heterogeneity and human ecosystems can be achieved in urban ecosystem designs for specific localities when tied to larger-scale ecosystem understanding. The neighborhood design strategies for the Baltimore Ecosystem Study can be scaled up through analysis of land-cover patch change from the preferred design changes. HERCULES, the High-Resolution Classification of Urban Land Cover and Environmental Systems (Cadenasso et al., 2007a), allows for such integration through small scale changes occurring over large areas. This system shifts the interest of master planning in zoning and land use, to contemporary ecology's framework of patch dynamics and urban design (McGrath et al., 2007). This system also marks a shift from sustainability motivated master planning's interest in codified best management practices (BMPs) to patch dynamics and urban design. Codified BMPs may represent a premature putting in order as they are based on the success of prior case histories, on insights from general ecological principles and assumptions rather than data concerning ecological function. For these reasons they may not be applicable in novel ecological circumstances. The changing forms of cities throughout the world coupled with climate change are in fact generating novel ecological situations for urban systems.

Back in New York, urban ecosystem research is accompanying a new comprehensive plan to accommodate one million new residents in an aging city while achieving new sustainability initiatives. The inheritors of Olmsted's and Moses' parks legacy, the NYC Department of Parks and Recreation, have expanded the notion of urban nature beyond the boundaries of parks. In the million trees program, one hundred thousand trees will be planted each year over the next 10 years. Each planting plot is georeferenced as a research site on tree health and mortality, social stewardship, and urban design factors. This integrated approach will assure a new understanding of the fine grain relationships of Jacobs' close knit neighborhood life inhabiting and observing the urban forest. In another initiative New York City Department of City Planning (DCP) under the directorship of Amanda Burden has fine tuned a new patchwork approach to zoning. Initially introduced by Mayor John Lindsay's Urban Design Group in 1964, Special District zoning abandoned master planning on a citywide, regional scale (Shane, 2005). This next generation of Special District Zoning was tested in the Long Island City Mixed-Use Special District and presaged similar strategies in Greenpoint-Williamsburg, East Harlem, and Chelsea. It is a targeted approach that encompassing micropatches of upzoning, e.g., from residential to commercial uses, downzoning in the reverse direction, mixed-use, and historic and industrial preservation (Shane, 2005).

Neighborhood design strategies in Baltimore, and a "million tree" research sites and patchwork zoning in New York can enable distributed and bottom-up adaptation to disturbances such as rapid climate change, rapid transition away from an oil based economy and immigration fluctuation. Together these design modes engage an image of the city as an adaptive system that is based on ways that everyday decisions have large effects. This is not a city where the scientist as a public figure offers controversial displaced points of view of the whole that motivates environmentalists and policy makers to make change based on fear. It is an attuned city where forestry, gardening, lifestyle, architecture, energy, circulation, cooking, and policy are understood as urban design innovations that are continuously modified in iterative consultation with ecologists, social scientists and policy makers. In other words, according to Latour (2004: 139) the scientist as public figure is valued for "their ability to provide frameworks, models, instruments and equipment, their capacity to record and listen to the swarming of different imperceptible propositions that demand to be taken into account", as well as "their ability to zoom in on an order of preference going from large to small that unblocks the situation by shifting the weight of the necessary compromises to other beings and other properties". This new view is exemplified by the study of nitrogen in Baltimore, and the greening stewardship and neighborhood resilience in NYC. The Commissioners' grid is an expression of the era of the sanitary city. Plans were put in place to provide clean water, efficiently remove waste and stormwater, and provide healthful living conditions (Melosi, 2000). Yet as the human population of the globe becomes more urban, with growing impacts on life support systems nearby and at a distance, concern must shift from sanitation to sustainability. Urban sustainability requires a reduction in the stress and reliance on engineering alone, while simultaneously enhancing the capacity of urban ecosystems to provide services locally and downstream.

Transitioning from sanitary to sustainable cities is a fundamental challenge that will require sustained effort and collaboration among researchers, policy makers and community stakeholders. Interdisciplinary research on urban system dynamics and on the myriad of social, economic, institutional, geographic and biophysical processes that influence the evolution of urban areas is a critical component. While we know something about the spatial heterogeneity that characterizes human and biophysical components of urban areas and the patch dynamics of some urban ecological and social-ecological processes, we lack an integrative understanding of how these spatially heterogeneous processes interact with each other across multiple spatial and temporal scales. Understanding how such cross-scale interactions and fast-slow dynamics influence the dynamics of urban systems is critical for achieving resilient urban futures. Resilience, defined both in terms of the amount of disturbance a system can absorb and still remain within the same domain of attraction and the adaptive capacity of a system in the face of small, constant or sudden change (Folke, 2006; Carpenter et al., 2001), is a characteristic of complex, adaptive systems and a key component of urban sustainability (Pickett et al., 2004). How to model resilience, as a part of the broad concern with sustainability, occupies the next section.

10.1. Modeling resilience

Understanding resilience requires a spatially explicit, dynamic modeling approach that can account for non-linear dynamics and cross-scale spatial and temporal interactions, key features of complex systems. Multiscale data analysis of key urban structure variables, e.g., population, income, employment and land-use variables, defined at relatively small spatial (e.g., Census block or block group) and temporal (e.g., annual) scales and over long time periods is a critical first step in modeling the complexity of urban systems. The increasing availability of electronic, spatial data on population, land use and firm location at a micro-scale offers new potential for better understanding complex dynamics (Batty, 2005) and some progress has been made in this regard. For example, Rozenfeld et al. (2008) use a spatial clustering algorithm with micro-scale population data that accounts for correlations in localized population growth across space at various scales and tests for significant differences in the parameters across scales.

Data analysis over a broader range of spatial scales and longer temporal scales than has previously been done is critical to understand the processes that underlie complex dynamics (Folke, 2006; Cadenasso et al., 2006b). However, inductive approaches alone are insufficient for modeling resilience. If urban systems are indeed path dependent, then qualitatively new dynamics can emerge in response to stochastic shocks or threshold effects. Assessing the resilience of any particular state of the system and comparing resilience of various states requires an understanding of not just the observed dynamics, but the range of possible dynamics that the system could exhibit under various conditions. This implies that models must go beyond predicting future outcomes based on historical data to a characterization of system-level dynamics for the full range of plausible values of key parameters. For example, in assessing the resilience of the City of New Orleans, it is not enough to identify a systematic relationship between income inequality and the City's resilience, e.g., as measured by its ability to successfully adapt after the sudden shock of Hurricanes Katrina and Rita. One would also like to know how a hypothetical change in the poverty rate or in the spatial distribution of poverty would affect its resilience.

Following Irwin et al. (2008), we propose that multiscale data analysis should be pursued in combination with process-based models of individual-level behavior and interactions that explain the emergence of systematic higher-scale patterns and system dynamics models that facilitate an exploration of dynamics across the full range of parameter space. This latter point is particularly important for understanding resilience. We argue that modeling urban resilience requires not only careful analysis of what is and what was, but also a systematic means of characterizing what could be or what could have been. To say that a city is more or less resilient requires a point of comparison: more or less resilient to what? Clearly the comparison is to other states of the system, but given the path dependent nature of urban systems, many of these states are not observable. Thus a modeling approach that can more fully represent these hypothetical states and assess the resilience of observed and hypothetical states of the system is needed. System dynamics models offer a means of doing so by summarizing the dynamics of the system for the full range of parameter values, both observed and potential.

System dynamics models are more comprehensive than empirical models and are needed for understanding long-term dynamics, but the necessary abstraction from empirical reality is also a danger of such models. This underscores the importance of grounding their development with extensive data analysis that can establish a set of empirical regularities with which the systems dynamic model can be validated by testing the model's ability to predict these observed outcomes. Data on one or multiple urban regions over a very long time period increases the number of possible outcomes or states of the system that are actually observed and thus increases confidence in the range of possible dynamics described by the model. Therefore, while data analysis alone is insufficient for understanding resilience and modeling long-term system dynamics, the development of spatially-explicit data on key urban variables over long time periods is essential.

11. Conclusions

The human habitation of the Earth has shifted from predominantly rural to majority urban over the last century. Although this new kind of anthropogenic biome, or anthrome (Ellis and Ramankutty, 2008), has been neglected by mainstream ecology for most of that same period, there were important contributions toward understanding urban ecosystems during that time. As the reality of the urban signature on metropolitan regions, distant hinterlands, and even the entire biosphere has become clearer, however, more and more ecological effort has been devoted to urban research. This realization has been accompanied in several countries by increased and long-term investment in urban ecological research, and indeed, in integrated socio-ecological research (McDonnell et al., 2009). Therefore, the recent advances in urban ecology have been swift and multifaceted. We summarize some of the key attributes, findings, and needs of contemporary urban socio-ecological research discussed above.

An important finding of urban ecological research, both that focused on patches within urban complexes and on the structure and function of entire conurbations, is that ecological function still persists. This is against the assumptions traditionally made in many disciplines of social science, geography, and economics, but in line with some pioneering work in these disciplines. A high retention of limiting nutrients such as nitrogen, and a capacity to contribute to carbon sequestration are examples of persistent ecological functions in cities. Ongoing long-term research is an important strategy to unravel the complexity of integrated socio-economic and ecological functions in cities, suburbs, and their remote exurbs, and to further evaluate the commonness of ecological functions in them.

Recognizing the ecological functioning within the boundaries of urban complexes has several benefits. First, by exploiting urban ecological functioning to process wastes, slow the flow of water, mitigate climate, and promote health and human wellbeing, quality of life in cities and quality of environment downstream can both be enhanced. Economic savings can demonstrably result from exploiting the ecological functioning that does or can take place in urban complexes. Second, the retained or restored ecological structures and functions in urban systems provide a connection of humans to nature. By nature we mean organisms, structures, and functions that reflect their own evolutionary history, inborn behaviors, and the working out of unmanaged interactions. Such wild or semi-wild components of urban complexes have their own intrinsic value, of course, but they are also the raw material for outdoor education, unstructured play and exploration, and a spiritual connection. Maintaining or restoring such wild places, in both small and large parcels, is a priority for the sustainable city of the 21st century.

Over the past decade or so, socio-ecological conceptions of the urban, and ways of modeling them have become more refined and usefully differentiated. For example, rather than using only traditional socio-demographic variables that emerged from the theory of cities as industrial production engines, new social and demographic concepts have emerged. While there remains a place for standard variables such as human population density, age structure, ethnic composition, and measures of educational attainment and wealth, contemporary social science has added households, lifestyle niches, and property regimes to the roster of variables that have explanatory and predictive power in linking biophysical and social causes and effects. In this new paradigm, a guiding question becomes, what are the biophysical and social allocation mechanisms for resources and justice.

The reciprocal relationship between socio-economic and biophysical processes in and near urban complexes often can be focused through one or more critical lenses. Land change science is a rubric that has emerged to signal the importance of land as the foundation for both social and biophysical processes. Land may in fact stand for the more complete concept of the human ecosystem in this theory. Both land use and land cover are addressed within this framework. Refined land-cover classifications, high-resolution social data, and an exploitation of the parcelized nature of urban complexes are components of advance in land change science.

Two aspects of urban ecosystems stand in close relation to land change – vegetation and water. Vegetation is one of the three main spatial structuring elements of urban areas, and its relationship to the remaining two – buildings and surfaces – is subject to policy, regulation, preference, and management resulting from individual, household, community, and government actions. Vegetation is a biocomplex parameter because it also reflects unmanaged succession, interaction with pollination vectors, dispersers, and climate, for example. Municipal interest in vegetation has become a new policy and management priority as many cities in mesic climates have established substantial programs to increase their tree canopy coverage. Of course, cities in more stressed environments, such as arid zones or coastal deltas, also desire to become more ecological. However, new environmentally appropriate models for "greening" these cities will have to be found that account for the prevailing environmental stresses or their economically and ecologically appropriate mitigation. Water is likewise a feature of urban systems that can serve as a node for integrated measurement, policy concern, and management. As drought is increasingly seen to be a problem in all the Earth's biomes and anthromes, the centrality of water in urban ecology is clearly justified. Sustainability has become a concern of many cities, and vegetation and water will play key roles in advancing this goal.

Our survey of advances in urban socio-ecological research indicates some conspicuous research needs. This list is not exhaustive. Urban ecological research on climate suggests that it is both a leading indicator of global changes that will have force more widely, and a problem requiring local mitigation and social adaptation. The generalizations about urban heat island and urban hydrological drought are two aspects of urban ecology that are well established. However, more fine scale focus is required to identify issues of distribution within urban complexes, and inequitable distribution of risk and adaptation to heat and water stresses of urban climate.

In promoting research on urban hydrology, increasing concern with fine scale budgets, pathways, wet spots, and artificially dryer patches is required. The hydrological connectivity in urban areas is enhanced in some places by outfalls from surface and subsurface drainage infrastructure, while other places are disconnected from hydrological recharge as a direct or indirect result of that same infrastructure. More fine scale measurements of soil water, ground water, and rainfall in and around cities are a research need. A conceptual framework to address both the longitudinal and the lateral dynamics in urban stream and water infrastructure networks and their watersheds is needed. Something akin to the stream continuum concept (Vannote et al., 1980) adjusted for urban realities may be useful. Whether nutrients and contaminants are processed or transported in urban stream/drain networks is a question for further research. Transport of nutrients in particulate as well as dissolved forms must be more widely addressed. Finally, assessing the content and effects of pharmaceuticals and personal care products in urban streams and receiving waters should be a more active area in the future.

How generalizable across the wide range of geographies, topographies, human densities, histories, cultural, and political contexts are the findings about urban socio-ecology? This must remain an open question for now. However, some seemingly robust generalizations are emerging, as described earlier in this paper. Several syndromes have been identified for urban ecological parameters. One, the urban stream syndrome comprises generalizations about channel incision, flash flooding, chemical compositions, and biotic properties. Another, the urban wildlife syndrome, comprises generalizations about food subsidies, the roles of generalist and invasive species, and the alteration of food webs. Faunistic homogenization has been noted across cities. In addition, insights about behavioral adjustments and genetic differentiation in urban populations are emerging. Plants exhibit several similar trends in terms of exotic species and generalist adaptation. Both the floras and faunas, especially the more mobile members, exhibit moderate to high degrees of homogenization across many cities measured to date.

Ecological homogenization is also seen in soil moisture, carbon budgets, and soil organic matter. The large nitrogen inputs associated with automobile use in cities contributes to uniformly increased nitrogen turnover in many urban studies. These emerging generalizations require examination across the wide range of life zones that cities represent. Zipf's power law of the distribution of city size also suggests a useful framework for comparison when temporal and regional contingencies are accounted for (Bessey, 2002).

A final point with policy relevance is the degree to which the scale of planning and urban design matches the scale of ecological functions within urban complexes. The absence of detailed ecological data within urban complexes coincided with the rise of master planning, a top-down approach to urban areas that seemingly assumed an absence of ecological functionality in urban areas. A counter to this was the European biotope approach, which identified ecological planning as a goal for cities. Biotopes were specifically classified to focus on clearly ecological components of cities. In the United States a counter was the inclusion of ecological, cultural, and historic layers in master plans. The reaction against master planning in the United States predated the rise of a multiscalar, functional, and integrated socio-ecological approach to urban complexes. However, that fine scale planning approach, usually identified as urban design, fits well with the opening of the opaque box of mid 20th century urban ecology. Urban areas are now conceptualized as internally heterogeneous, functionally differentiated, and highly interconnected ecosystem mosaics. Fine scale urban design operates at the same scale that modern ecosystem and landscape ecology have examined urban complexes. Thus, ecology has much to learn by measuring the effects of developments as though they were experiments (Felson and Pickett, 2005). The linked interactions of social and biophysical spatial heterogeneities at the fine scale can be exploited and applied through the practice of a fine scaled urban design approach that is also well connected with regional, watershed, airshed, and social connections. In this way, the growing concern of science and society with sustainability can be enhanced and applied.

Acknowledgements

The preparation of this review and the research of the authors into the structure and function of urban socio-ecological systems was supported by the National Science Foundation's Long-Term Ecological Research program. In particular the authors acknowledge support by NSF grants DEB-0423476 and Biocomplexity in Coupled Natural-Human Systems program (BCS-0508054). The USDA Forest Service contributes substantially to research in Baltimore from which we draw here. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the authors and do not necessarily reflect the views of the National Science Foundation. STAP thanks the Office of the Provost for an M.L. King, Jr., Fellowship, and the Department of Urban Studies and Planning at the Massachusetts Institute of Technology for providing a stimulating intellectual environment in which this paper was completed.

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