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Chapter 6

Biodiversity on the Urban Landscape

Katalin Szlavecz, Paige Warren, and Steward Pickett

6.1 Introduction

Expanding urbanization is one of the leading types of land use change today. In 2005, 49.2% of the world population lived in cities, and this number is expected to reach 60% by 2030 (United Nations Population Fund 2007). Urban population in the US is already above 70%, while in the developed countries in general, urban population accounts for more than 80% of national totals (Table 6.1). Even more important for biodiversity is the rate of change of urban and suburban land covers. On a regional basis, the rate of land conversion to urban uses, in the broad sense, often exceeds the population growth in that same region. For example, from 1982 to 1997, developed land (according to the U.S. Department of Agriculture's Natural Resource Conservation Service, land is classified as one of the several land cover/use categories, either urban or other built-up areas, or rural transportation land) in the 48 contiguous United States increased by 34%, while during the same period, population increased by only 15% (USDA NRCS 2001; US Census Bureau 2000). Conversion of land from agricultural and wild categories to the general category of urbanized uses was thus more than twice as fast as population growth for the same 15-year period. Such changes are quite significant for biodiversity (Forester and Machlis 1996).

Cities – a term we will often use as shorthand for the broader array of urbanized areas, from central business districts to old residential areas, to commercial and industrial sites, to new suburbs, as well as the new edge cities and exurban fringe – affect biodiversity because they present unique habitats. Cities are densely populated, highly modified systems resulting from destruction, alteration, and fragmentation of the original wildland or older rural habitats and from creation of new habitat types. Built structures and impervious surfaces make up a large percentage of urban land cover, while remnants of original habitats may still exist. In addition, “volunteer” or semiwildlands are important in some urban areas. Urban landscape is a patchwork of many land uses which, along with altered hydrology (Paul and Meyer 2001; Groffman et al. 2003) and climate (Botkin and Beveridge 1997; Brazel et al. 2000), profoundly affect biodiversity at all spatial scales (Sukopp and Starfinger 1999). Many elements of this landscape are heavily managed by

Table 6.1 Trends of urbanization by major areas

Percentage of population residing in urban areas				Projected annual rate of urbanization (%) 2005–2030
	1950	2000	2030	
Africa	14.7	36.2	50.7	1.12
Asia	16.8	37.1	54.1	1.23
Europe	50.5	71.1	78.3	0.33
Latin America and the Caribbean	42.0	75.4	84.3	0.34
North America	63.9	79.1	86.7	0.29
Oceania	62.0	70.5	73.8	0.17
<i>World</i>	<i>29.0</i>	<i>46.7</i>	<i>59.9</i>	<i>0.83</i>

Source: United Nations, Department of Economic and Social Affairs, Population Division 2006. Used with permission.

humans, and direct management and its indirect effects constitute major forces shaping diversity in cities. Clearly, the assembly rules driving biological community structure in cities are very different from those driving *natural*, less human-dominated communities.

Biodiversity has different meanings to biologists, to policymakers, and to the public. Biologically, the term applies to many levels of biological organization (Noss 1990). Components of biodiversity include genetic diversity, species richness, and landscape diversity. Conservation biology often emphasizes the number of rare or endemic species within a community rather than simply focusing on the number of species of an area. However, most urban biodiversity studies focus on species richness indexed by the number of species in a given area, which is commonly called alpha-diversity (Magurran 1988). Another important metric of diversity, species evenness, is often reported as well. The focus of the following discussion is on species richness and species composition on the scale of habitats within urban areas.

6.2 Some Key Characteristics of Urban Ecosystems

As perhaps the most human-dominated system on the planet, cities represent a setting in which the effects of human demography on biodiversity may be most evident. Several human demographic trends are known to contribute to the impacts of urban areas on biodiversity. Increased urbanization – that is, increased population size of urban areas, which on average equals about 1.5 times the US national level of population growth – is due to both increases in the resident urban population and immigration from rural areas and abroad (Dow 2000; Cincotta et al. 2003, p. 53). Moreover, the area of most cities is expanding faster than their populations, a phenomenon known as urban sprawl (e.g., Alberti et al. 2003; Radeloff et al. 2005). This is due in part to shrinking household sizes (Liu et al. 2003) but also to larger parcel sizes in newer suburbs compared to older suburbs or central cities (Heimlich and Anderson 2001). The resulting conversion of wild or rural lands to urban lands

generally produces reduced diversity of native flora and fauna and elevated numbers of exotic species (Kowarik 1995; Marzluff 2001; McKinney 2002), but there are exceptions to this pattern (Davis 1999; Samu and Szinetár 2000; Niemeleä et al. 2002). Furthermore, the kinds of human effects on biotic communities are far more complex than broad elimination of populations of native species or native habitat. Humans actively create biological communities in their parks, gardens, institutional grounds, and yards. The characteristics of these constructed communities depend on choices made by organizations, communities of people, households, and individuals (Odum 1970; Whitney and Adams 1980; Hope et al. 2003; Martin et al. 2004; Kinzig et al. 2005; Grove et al. 2006b). These agents and their decisions are in turn embedded in cultural traditions and socioeconomic networks (Machlis et al. 1997; Pickett et al. 1997). Thus, to understand the human drivers of the patterns of biodiversity in urban areas, we must find ways to integrate social science approaches with conventional ecological approaches to understanding biological communities (Cadenasso et al. 2006).

As humans actively construct biological communities in cities, they may juxtapose species that evolved on different continents and under different biophysical conditions (Hobbs et al. 2006). These novel communities often simultaneously have more *total* species but fewer *native* species than the surrounding native habitat (e.g., Marzluff 2001; McKinney 2002). The food resources provided by this novel habitat may actually be enriched relative to nonurban habitat, especially in temperate regions. For example, there may be a greater numbers of fleshy-fruited plants in urban sites (Beissinger and Osborne 1982). These drastic rearrangements of flora and fauna are thought by some to be leading toward a global homogenization of biotic communities and consequently a total reduction in global biological diversity (McKinney and Lockwood 1999; Blair 2001; Pouyat et al. 2006; Schwartz et al. 2006). Regardless of whether this is true, the novel habitats created by humans clearly shape urban patterns of biodiversity. Urban flora and fauna are different from those of the surrounding areas.

Several factors account for the differences between urban and nonurban species assemblages. The modified urban environment may be suitable only to a subset of the original flora or fauna. This mainly depends on the ecological requirements of the species in question. However, while urban environment can be stressful for some species due to pollution, habitat fragmentation, etc., others may thrive in the cities because humans create favorable microhabitats or abundant resources for them. An example is the increased number of vine species present in forest canopy gaps in Baltimore, MD as compared to the smaller roster in rural forest gaps (Thompson 1999). These species are often nonnatives that have been associated with settlements and human activities for a long time, and which have been widely dispersed by people, or which readily spread on their own.

Other modified habitats also occur in urban areas in large numbers and include greenhouses, basements, compost piles, and green roofs. Many species have been described and are still known only from greenhouses (Korsós et al. 2002). Others, although first described from greenhouses, later spread into outdoor environments. One example is the common pillbug, *Armadillidium vulgare* Latr., in North America.

Using a combination of historical data and molecular approaches, Garthwaite et al. (1995) showed that *A. vulgare* on the east coast of the US was first reported from greenhouses but then subsequently was found in the south and west. Moreover, the west coast populations of *A. vulgare* are more similar to the Mediterranean populations in Europe, whereas the east coast populations are more similar to Atlantic European populations. This difference indicates independent introductions by different cultures on the two coasts, a pattern that corresponds well to the human immigration history of North America.

Urban environmental change is rapid and was until recently considered too fast for animals and plants to adapt. However, genetic evolution is a documented result of urbanization. On the Pacific coast of the United States, evolved changes in tail color were found in a population of birds in San Diego, CA (Yeh 2004). In plants, heavy metal tolerance in urban microhabitats or *brownfields* is an example of rapid evolution (e.g., Velguth and White 1998). Some species can also adapt nongenetically, that is, behaviorally or culturally, to the changes wrought by urbanization (Boyd and Richerson 1985; Yeh and Price 2004; Parker and Nilon 2008). For example, some birds have been documented to alter their songs in response to urban noise (Slabbekoorn and Peet 2003; Warren et al. 2006). Although these examples of populations adapting to the changing urban conditions join those of genetic change following industrialization (e.g., the textbook case of the peppered moth, *Biston betularia*), some species might disappear because they cannot adapt rapidly enough (genetically or behaviorally) to the novel conditions in an urban setting (Slabbekoorn and Peet 2003; Shochat et al. 2006; Warren et al. 2006).

6.3 Why Study Urban Biodiversity?

6.3.1 Values of Biodiversity

The reasons to study biodiversity in urban areas are many. Perhaps the most obvious reason is an esthetic or ethical one. Humans are attracted to nature and its living creatures. E.O. Wilson called this phenomenon “biophilia” and defined it as our “innate tendency to affiliate with life and lifelike processes” (Wilson 1984). Being surrounded by plants and animals creates a sense of peace and tranquility (Coley et al. 1997; Frumkin 2001). Given a choice, in the city people may prefer to live near a park, or have a view to a lake or river, assuming the social context and perceptions of hazards to be equivalent near such amenities and elsewhere. We surround ourselves with plants in our apartments or balconies (Rapoport 1993). We keep pets; we find having an aquarium in the living room relaxing. We plant shrubs and trees even in the tiniest yards and welcome birds with feeders in the winter. It does not matter whether these species have an ecological function (most of the time they do) or *just* esthetic value. If the birds help control harmful insects throughout spring and summer, that is an extra benefit; we just like having them around.

We liked caring for our pets even before scientific studies demonstrated that they lower blood pressure.

In spite of the profound esthetic values of urban biota, there are also many practical values or *ecosystem services* that are provided by urban biodiversity. Trees cool local climate, and together with herbaceous vegetation they take up excess nutrients and reduce runoff, but most people are unaware of these facts (McPherson et al. 1997). Community gardens utilize vacant spaces in a unique way. In addition to growing vegetables or flowers, they provide a meeting place for the neighborhood and promote social interaction (Burch and Grove 1993). They add to the city's green spaces and, by attracting pollinators and nectar feeders, they help maintain biodiversity (McIntyre and Hostetler 2001). Green roofs reduce storm water runoff, regulate building temperatures – thus conserving energy, and increase wildlife habitat area (Oberndorfer et al. 2007).

6.3.2 *Roles of Exotic Species*

In spite of the variety of values associated with urban vegetation, exotic species are often excluded from such recognition. The simplistic view that “exotics are always bad” needs revision. Exotics can be valuable in several ways. They may serve as important resources for native species. For example, in Davis, CA, 29 of 32 native butterflies breed on nonnative plants, many commonly designated as *weeds* (Shapiro 2002). Many of the native host plants no longer occur in the region, and exotics have taken on some of their important ecological roles. For example, various species of tamarisk (*Tamarix* spp.) have become important nesting sites for an endangered bird species. Many exotic species have existed in their nonnative host ecosystems for centuries and have integrated into the ecosystems to the extent that it is almost impossible to determine what the system must have been like before their arrival (Sukopp et al. 1990). Indeed, to return the biotic components of urban ecosystems to some ideal, pristine condition would be quite impossible in almost all cases of long residency.

A very special subset of exotic urban biodiversity is represented by species inhabiting artificial, human-made environments such as greenhouses and botanical gardens. As a result of the plant trade and the equable environment, the soil invertebrate species composition of some greenhouses is beyond imagination. For instance, a survey of only a few greenhouses in Hungary resulted in soil invertebrates originating from Asia, South America, and Africa (Csuzdi et al. 2007; Vilisics and Hornung 2009; Table 6.2). In general, the presence of such introduced species assemblages in greenhouses and similar novel environments are considered by ecologists to be undesirable, because they increase the chances for wider exotic introductions for which the ultimate consequences to native species and communities are yet unknown.

Taxonomists note that some species are only known from greenhouses (e.g., Korsós et al. 2002) or other human-engineered environments. The region of their

Table 6.2 Soil invertebrates and their known or possible zoogeographical origin from three greenhouses in Hungary

	Origin	Percentage of fauna of Hungary
Earthworms		
<i>Phitamera bicincta</i> (Perrier, 1875)	SE Asia	7.8
<i>Amyntas corticis</i> (Kinberg, 1867)	SE Asia	
<i>Eudrilus eugeniae</i> (Kinberg, 1867)	West Africa	
<i>Ocnodrilus occidentalis</i> (Eisen, 1867)	Central America?	
<i>Dendrobaena attemsi</i> (Michaelsen, 1902)	Alps	
<i>Microscolex phosphoreus</i> (Dugés, 1837)	S America?	
<i>Dichogaster bolau</i> (Michaelsen, 1891)	E Africa	
Isopods		
<i>Armadillidium nasatum</i> (Budde-Lund, 1885)	Atlanto-Mediterranean	7.3
<i>Cordioniscus stebbingi</i> (Patience, 1907)	Iberian	
<i>Reductoniscus costulatus</i> (Kesselyák, 1930)	SE Asia	
<i>Trichorhina tomentosa</i> (Budde-Lund, 1893)	Tropical America	
Millipedes		
<i>Choneiulus palmatus</i> (Nemec, 1895)		5
<i>Cylindroiulus truncorum</i> (Silvestri, 1896)	N. Africa	
<i>Amphitomeus attemsi</i> (Schubart, 1934)	S America	
<i>Poratia digitata</i> (Porat, 1889)	S America	
<i>Cynedesmus formicola</i> (Cook, 1896)	Canary Islands	

biogeographical origin is not known. Indeed, these species may no longer exist elsewhere. A few years ago a small centipede, *Nannarrup hoffmanni*, made headlines in the New York Times after being identified in the leaf litter in New York City's Central Park. It turned out to be a unique, newly described species whose discovery generated great excitement among taxonomists (Foddai et al. 2003). At present, Central Park is its only known locality. Soil and leaf litter generally harbor diverse communities, with many species undoubtedly awaiting description.

6.3.3 Contribution of Urban Studies to General Scientific Inquiry

The altered conditions and relatively fast rate of change in cities provide the basis for *natural experiments*, with rural or wild environments serving as controls, or more properly, reference systems. This situation provides urban ecologists with opportunities to observe and compare phenomena at the organismal, population, community, and ecosystem realms, and test general ecological hypotheses. An example is the relationship between disturbance and species diversity, which today goes beyond testing the intermediate disturbance hypothesis (Connell 1978). Going beyond intermediate disturbance requires identifying mechanisms by which potential invaders respond to specific human actions (Bart and Hartman 2000). This refinement examines specific events and responses rather than treating disturbance as a highly aggregated and hence inconsistent collection of diverse events.

In addition to serving as convenient experimental substrates, urban areas can serve as models of global change (Carreiro and Tripler 2005). The physical environment in cities, which includes elevated CO₂ concentrations, higher temperatures, and altered hydrological cycles, mimics key components of global climate change, thus providing opportunities to study responses of biota to such changes in existing rather than simulated environments. One such study in the Baltimore Metropolitan Region is underway (Ziska et al. 2004). Under these altered climatic conditions, shifts in species composition from natives to nonnatives, and from specialists to generalists, help to understand how redundancy may function in biological systems elsewhere as climate changes.

6.3.4 *Urban Ecosystems and Biodiversity Education*

Urban areas can serve as important venues for ecological and environmental education (Berkowitz et al. 2003). For most people, the only real encounter with the diversity of life happens in their city backyard or their suburban neighborhood. We have an obligation to help children and adults to learn about the species surrounding them and the role they play in that ecosystem (Miller 2006).

An example of engaging the public while gathering useful information on urban biodiversity is BioBlitz. The idea behind BioBlitz (<http://www.pwrc.usgs.gov/blitz/>) is to bring together taxonomists, park managers and the public as volunteer individuals, families, or school groups to document the biodiversity in their immediate environment. BioBlitz in cities usually takes place in urban parks and is a concentrated effort for a short period of time. Over a 24-h period, organisms are collected or observed and recorded for as many taxonomical groups as possible. It is not intended to be a rigorous scientific biodiversity survey; rather, it can serve as a starting point for scientific assessments. Since 1996, over 100 BioBlitz sites have been established all over the world. In addition to documenting species present in an area, the involvement of the public in such a fun event is a true benefit of this activity. People go home with a greater knowledge and appreciation of their own surroundings.

In other ongoing citizen science programs, instructions are given on websites along with simple taxonomic keys. Alternatively, schools or citizens are asked to send in specimens, and the resulting data are entered into a central database. The Wormwatch Program in Canada (<http://www.naturewatch.ca/english/wormwatch/>) and Walking with Woodlice in the UK (<http://www.nhm.ac.uk/woodlice/>) are examples. Survival and reproductive success of common backyard birds are the focus of the Nestwatch Program in the United States. (Marra and Reitsma 2001; Evans et al. 2004). DC Birdscape, though a coordinated efforts of several agencies and volunteers, systematically counts the birds in the Washington, DC area (Hadidian et al. 1997). Awareness and appreciation of urban biodiversity enhances the quality of life facilitating conservation efforts outside of the city as well (Savard et al. 2000) (Figs. 6.1 and 6.2).

Fig. 6.1 In the absence of large urban green spaces constructed plant assemblages, such green roofs contribute to human well-being, as well as maintaining higher biodiversity. (Photo was taken in Munich, Germany by K Szlavecz)



Fig. 6.2 Students sampling soil invertebrates in an urban forest in Baltimore, MD. Active participation in biodiversity surveys increases awareness and appreciation for the variety of life. (Photo was taken in Baltimore by K Szlavecz)

6.4 Research Approaches to Urban Biodiversity

Research on urban biodiversity must account for the spatial heterogeneity and complexity of urban ecosystems. The urban landscape is highly heterogeneous and exhibits striking changes from the rural surrounding to the urban core. Patchiness and the urban–rural gradient have been the two major factors guiding biodiversity assessments. These two perspectives account for the spatially complex nature of urban systems (Figs. 6.3 and 6.4).

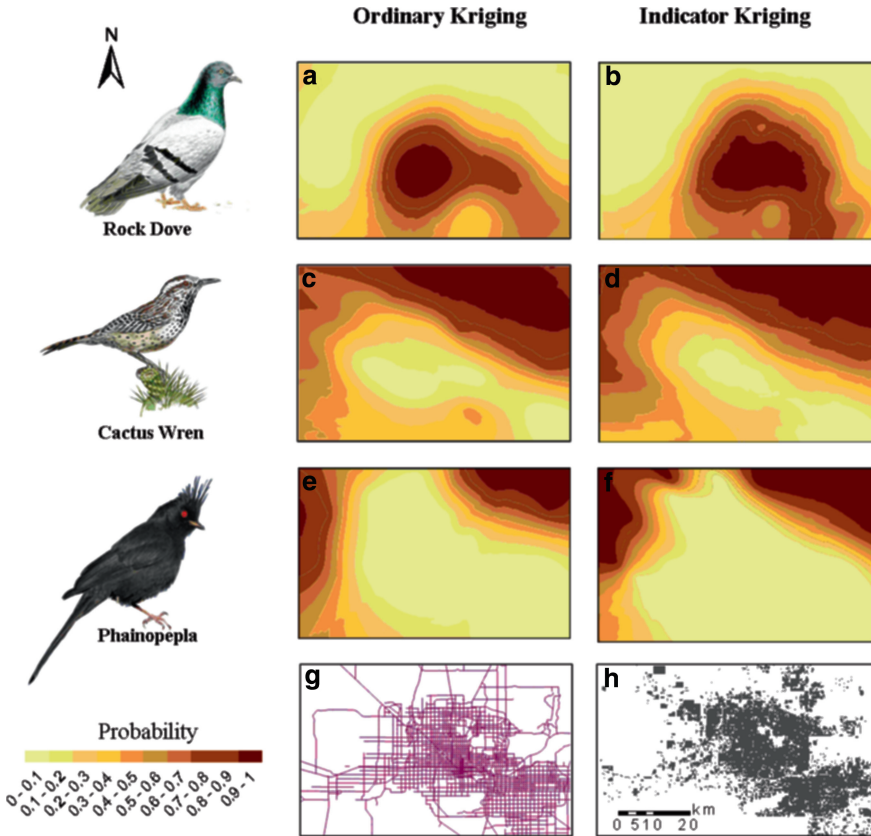


Fig. 6.3 Probability maps of the avian species analyzed using two kriging methods (a–f), major road (>3 lanes) networks (g) and urban land use (h) as defined by the Maricopa Association of Governments (2000). The stated probability indicates the likelihood that the number of individuals of a species will exceed the observed median. Three ecologically unique species are presented. Rock doves represent a species that is strongly linked to the urban ecosystem. The other two are native species. *Phainopepla* is an abundant species that very rarely utilizes the urban ecosystem. Whereas, the cactus wren encroaches into the urban core, it is more prevalent in the desert. Ordinary kriging was conducted with transformed data in order to satisfy the assumptions of normality. Indicator kriging was conducted with untransformed data. Images of birds are courtesy of Global Institute for Sustainability, Tempe, AZ. (Figure reproduced with permission from Walker et al. 2008)

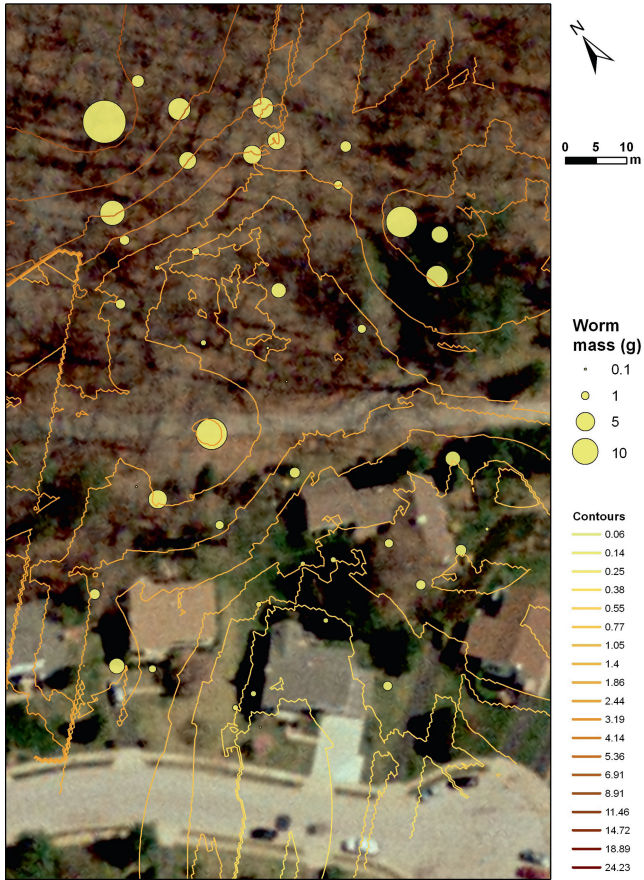


Fig. 6.4 Earthworm abundance and distribution on the heterogeneous urban landscape. The width of dots and contour lines indicate the number of earthworms collected from $25 \times 25 \text{ cm}^2$ quadrats. Data were collected in a typical suburban neighborhood, at the northern edge of Baltimore, Maryland. Data were compiled and the map was constructed by Erle Ellis, University of Maryland, Baltimore County

6.4.1 Urban–Rural Gradients

The urban–rural gradient approach is based on the assumption that key characteristics of the contrast between dense urban and the exurban fringe can be ordered conceptually. In rare cases, the ordering may be literal and follows a transect from the central business district to the rural fringe. In most cases, however, the complex mosaic of urban, suburban, exurban, and rural conditions are ordered in an abstract, multivariate space. Both of these approaches are captured within the concept of the *urban–rural gradient* (McDonnell et al. 1993). However, it is important to recognize that the conceptual gradient may not reflect any real transect on the ground in a

specific metropolitan area. This is due to the fact that contemporary urban mosaics are multicentered, spatially extensive, and highly networked by transportation and information infrastructure (Bruegman 2005). The traditional concept of a single urban core with gradual transitions to the resource-supplying hinterlands is not suitable to the twenty-first century urban situation in either industrial or developing nations (Garreau 1991). Transects and abstract gradients have been powerful tools for studying both the biophysical and social variation in urban structure and function (Cadenasso et al. 2006; Dow 2000).

6.4.2 Island Biogeography in Cities

An important approach to the study of urban biodiversity has been to apply island biogeography theory (MacArthur and Wilson 1963, 1967) to cities. Island biogeography was, of course, developed for oceanic islands where isolation, local extinction, and the supply of propagules from continental source areas were all keys. This theory was applied to continental situations, such as mountain tops and lakes, early in its history. Ultimately, it was also applied to green space islands in cities. Such green islands result from land clearing associated with urban development and suburban sprawl, which leads to fragmentation of the existing native habitats. These isolated patches, surrounded by a different type of land use can be considered as habitat islands and, hence, subject to island biogeography theory. Urban biodiversity studies often use the theory of island biogeography to test hypotheses about patterns of species richness in these remnant habitats.

Of course, urban habitat fragments are different from oceanic islands in many ways. Urban habitats are primarily nonequilibrium systems with altered regimes of biophysical disturbance agents and continuous anthropogenic influence. Fire suppression and flood control are two common examples of altered disturbance regimes. Humans may also actively manage or use these habitat fragments, for example, by cutting and removing vegetation and trapping unwanted mammals. Recreation or collecting flowers, nuts, and mushrooms are examples of activities in urban habitat islands, which may affect their biodiversity. Colonization and extinction in the habitat fragments are influenced by humans, as they deliberately or accidentally transport species among these patches or they introduce nonnative species into the surrounding region. Alteration of the habitat may encourage more generalist species leading to altered species interactions, modified community structure, and ultimately altered ecosystem function. In light of such a suite of direct and indirect influencing factors, it is not surprising that urban species-area relationships are typically more complex than the textbook examples of island biogeography.

Numerous studies tested species-area and species-distance predictions in urban areas. Higher species richness in larger fragments was reported by Frundt and Ruszkowski (1989), Sasvari (1984), Yamaguchi (2004), and Watson et al. (2005). Kitahara and Fuji (1997) successfully applied the theory of island biogeographical approach when analyzing the butterfly communities in newly designed parks

in Tsukuba City, Japan. Schaefer and Kock (1979) found that ground beetle (Carabidae) species richness correlated with structural diversity of green islands better than with the island area in Kiel, Germany. Plant species composition and vertical structure were the determining factors for bird species richness in similar sized urban woodlots (Tilghman 1987), while landscape matrix was an additional factor in Australian bird assemblage structure (Watson et al. 2005). While distance of forest fragments from the urban core correlated well with carrion beetle (Silphidae) richness in Baltimore, the area of the fragments did not (Wolf and Gibbs 2004). Arthropod species richness did not vary with fragments size in Sydney, Australia (Gibb and Hochuli 2002), but species composition did. Lack of independence of the above multiple factors may lead to inconclusive results (Whitmore et al. 2002).

6.4.3 Patch Dynamics

The theory of patch dynamics incorporates refinements beyond the assumptions of the classical theory of island biogeography and, hence, has an important role to play in stimulating and organizing urban biodiversity studies (Pickett and Rogers 1997). Patch dynamics relaxes the assumption that areas between islands, or more broadly, any patch of interest, is completely hostile to the survival or movement of potential colonists to the focal patch. The area outside the focal patch can be considered to be differentially suitable for the survival, movement, and activities of organisms that also live in that patch. Thus, the matrix is not a uniformly homogeneous and hostile soup but is itself a heterogeneous patchwork or field of gradients that differentially affect organism performance and ecosystem function. As mentioned earlier, urban areas are manifestly patchy across space due to social differentiation, economic investment, a hierarchy of transportation networks, contrasting built structures, soils, substrates, impervious surfaces, and the presence and activity of different plants, fauna, and microbes (Cadenasso et al. 2006) (Figs. 6.1 and 6.2). Therefore, studying the extensive urban ecosystem as though it were a mosaic of contrasting patches or a field of gradients is a powerful way to address complexity in a realistic way. Indeed, new urban classifications are aimed at quantifying and portraying this patchiness in ways that integrate both social and biophysical structures (Cadenasso et al. 2007).

The changes in urban systems are also captured in the theory of patch dynamics. Urban areas grow on their edges, as illustrated by earlier citation of data on the disproportionate amount of land that is being converted to urban uses compared to the growth of urban population. In many postindustrial cities, internal dynamics are also conspicuous, as old industrial sites, underutilized shopping districts, and old residential areas are thinned or entirely abandoned. Disinvestment and depopulation defines an internal “urban frontier” that is equally dynamic if less extensive (Burch and Grove 1993).

A common theme of the studies of the structure of urban biodiversity relative to urban structure, demography, and human decision making is the emphasis on biotic species composition over species richness alone. Knowledge of species identity in an assemblage is important, because it allows further community analysis, such as detection of guild structure and the ratio of generalists to specialists or natives to nonnatives. Furthermore, the focus on species composition pinpoints fragments with unique species assemblages, which function as areas of exceptional biogeographical value. All of these biotic details help to make better conservation decisions (Roy et al. 1999; Gibb and Hochuli 2002; Watts and Lariviere 2004; Samu and Szinetár 2000; Schwartz et al. 2002; Smith et al. 2006a, b).

6.4.4 *The Species Approach: Exotics and Introductions*

Species introduction in urban systems is common, and species-based research is an important approach to urban systems. Urban and suburban areas often have higher species richness than surrounding wildlands. However, this is due to the addition of nonnative species in urban sites (Pyšek 1993, Kowarik 1995). Some of these species, such as ornamental plants, are deliberately brought to the cities, while others are introduced accidentally. This second class is exemplified by many soil organisms brought over as stowaways in ship ballasts (Lindroth 1957). Exotic species introduction and invasion has become a global environmental and economic issue (Pimentel et al. 2000). Accidentally introduced species are often synanthropic – species that tend to be associated with humans or their activities. These species are often generalists, and they typically move along with human settlement. Moreover, cities are considered to be highly disturbed environments, which tend to favor colonization of *weedy* species.

The similarity of urban environments and the subset of synanthropic species that follows human movement results in similar flora and fauna in worldwide, a phenomenon called biotic homogenization (McKinney 2006; McKinney and Lockwood 2001). This ecological process is of a great concern in conservation biology, because urban environments can serve as jumping-off points for exotics that can subsequently colonize rural or wild areas, often outcompeting native species. In addition, exotics can become nuisance or pest species as rats, cockroaches, fire ants, and the Asian longhorn beetle. It is beyond the scope of this chapter to analyze the factors that make good colonizers, or which contribute to susceptibility of potentially invisable environments to the establishment of a new invader. However, urban environments have much to contribute to understanding these important questions, which should be incorporated into urban ecological research. Biotic homogenization is of concern because the local and regional pools of biodiversity represent genetic capital which can be spent in future bouts of evolutionary adaptation to changing environments.

6.4.5 *Human–Natural Coupling in Urban Systems*

A further important approach to urban ecosystems is interdisciplinary study that incorporates both human drivers of biotic change and the response of humans to ecological changes (Cadenasso et al. 2006; Pickett et al. 2008). Integrating the full range of possible human drivers to studies of urban biodiversity requires extensive collaboration and interaction between social and natural scientists (Kinzig et al. 2000; Redman et al. 2004). Machlis et al. (1997) developed a Human Ecosystem Framework to guide researchers in selecting variables to consider in their models. They advocate using multiple variables to describe each parameter. Forester and Machlis (1996) applied an early version of this approach to understanding biodiversity loss at the national level using existing datasets. There are many datasets available for identifying social characteristics of subunits of urban areas such as neighborhoods or other patches. These include US Census data or similar datasets for other countries (Hope et al. 2003; Melles 2005), marketing analyses such as Claritas (Weiss 2000; Grove et al. 2006a), and municipal datasets (Grove et al. 2006b). These examples offer a starting place for considering the role of demography and human institutions in urban ecosystems.

An example of how demographic and social data can be linked with biodiversity and ecosystem performance data appears in Troy et al. (2007). That study analyzed the presence of woody and grassy vegetation and expenditure on their management in terms of the Claritas market-segmentation data. The authors discovered that the potential to add new vegetation on residential parcels in the city of Baltimore, MD, which constitute 60% of the land surface, was best predicted by the market segmentation based on 15 categories. This level of aggregation includes basic demographic variables but also includes aspects of social stratification. The complete market segmentation, recognizing 62 lifestyle clusters, best predicts variation in the actual amount of woody or grassy vegetation planted on residential parcels. Specific variables related to the realized amount of vegetation include family size, presence of detached homes, and marriage rates. Both the potential and the realized vegetation cover were related to population density, housing vacancy, and housing age. The percentage of African–American residents related negatively to expenditures on yard-related expenses, though the causes are not yet known.

For a richer understanding of the role that human values and actions play in patterns of biodiversity, we will need to forge collaborations between social scientists and natural scientists. This is happening in many places, including at two urban Long-term Ecological Research (LTER) sites were funded by the National Science Foundation: the Baltimore Ecosystem Study (BES LTER) and Central Arizona-Phoenix (CAP LTER) (Pickett et al. 1997; Grimm et al. 2000; Cadenasso et al. 2006; Pickett et al. 2008). These two LTER studies and other LTER sites that incorporate human demography and activities in their regional scope are exploring new territory for ecology. Researchers in such projects examine biodiversity and the ecological function of biotic assemblages throughout extensive mosaics. Thus, they examine the ecology *of* the city, rather than focusing only on

the obvious green spaces. The more narrower approach can be labeled ecology *in* the city (Grimm et al. 2000). They examine the structure and dynamics of all sorts of patches in the urban mosaic. In addition, research teams jointly examine social and biophysical aspects of patch structure and function. Finally, partnerships with managers and policy makers ensure the relevance of surprisingly large proportions of such integrated research (Pickett et al. 2007).

One important component of how biodiversity functions in urban systems is especially lacking. Human values of biodiversity are little understood even in more natural environments. Some authors suggest that there are particular kinds of landscape preferences common to all humans (Oriens and Heerwagen 1992; Ulrich 1993). But there is also variation in human preferences. One recent study used mail surveys to measure variation in human values of birds and their participation in activities that either benefit (e.g., feeding) or harm (e.g., owning free-roaming cats) birds along an urban to rural gradient (Lepczyk et al. 2004). Clergeau et al. (2001) also explored variation human perceptions of birds along an urban–rural gradient in France. Since the diversity of human groups in close proximity to one another reaches its peak in urban settings, cities represent a natural laboratory for exploring variation in human values for particular kinds of organisms and biodiversity. Results to date show surprising similarities in the environmental perceptions and values of wealthy and disadvantaged residents of some urban areas (Vemuri et al 2009), while at the same time it is clear that environmental hazards and amenities are disproportionately visited on poor or minority residents compared to wealthy and majority persons (Boone 2002).

6.4.6 Ecosystem Function

Ecosystem or habitat function is an additional important approach to understanding urban systems. Because, as mentioned before, urban areas contain many novel habitats and unprecedented environmental conditions, understanding the function of these habitats becomes all the more critical. More studies need to incorporate a mechanistic approach to understanding urban biotic communities rather than simply documenting the presence and number of various species. When mechanistic studies have been undertaken, there are sometimes conflicting results. For example, some studies suggest that levels of predation on birds are lower in cities (reviewed in Shochat 2004), but others suggest that predators occur at even higher densities in the city than in the surrounding countryside (Sorace 2002). Studies of soil invertebrates have addressed mechanistic questions such as the effect of species composition on rates of decomposition (Broll and Keplin 1995) and biogeochemical cycling (Steinberg et al. 1997; Pavao-Zuckerman and Coleman 2005; Szlavecz et al. 2006), but there are far fewer studies of invertebrates than of the more charismatic species such as birds.

The ecosystem function of different species, especially exotics vs. natives, is a significant component of the functional urban approach. A specific example of such

research is a study by Groffman and Crawford (2003) in Baltimore. Sites dominated by the exotic *Ailanthus altissima* and *Acer platanoides* had higher levels of soil nitrate and nitrification rates than sites dominated by native trees. Thus, exotic species accelerate both nitrogen availability and loss. Sites dominated by the exotics trees also had higher soil moisture. It is not known whether the exotics colonize wetter sites or alter water balance in sites they invade.

Ehrenfeld (2003), in a wide-ranging literature review of the functions of exotic species in urban systems, concluded that invasive exotic plants often increase biomass and net primary production. Furthermore, exotics increase the availability and change fixation of nitrogen, resulting in litter of higher quality that tends to decompose faster than litter of native species. However, not all exotic species produce these results, with the opposite relationships sometimes found. There are no patterns in other components of nutrient dynamics such as soil pools of carbon and nitrogen. The same species can behave differently at different sites, suggesting that specific environmental conditions may determine the nature of ecosystem effects. Exotic plants differ from native species in terms of biomass, rates of productivity, tissue chemistry, plant morphology, and seasonal timing.

6.4.7 Overcoming Logistic Problems

Whatever research approaches are used in urban systems, there are logistical problems. The use of experiments is one of these approaches that are problematic in urban settings. To move beyond correlative studies, ecologists prefer to use manipulative field experiments with proper replication and control. However, unless the experiments are small scale, such an approach poses a problem in cities, due to highly parcelized land ownership, zoning regulations, access, and other issues (Cook et al. 2004). Scientists have to work closely with the city government and its departments, the residents and various organizations, and all of this puts constraints on ecological experiments. The experimental method itself, as mentioned before, is sometimes a constraint. Manipulating densities of exotic species to measure their effect on native species or on ecosystem function can be impractical, unpleasant (e.g., involving pest species), or simply intrusive. In addition, in order to conduct research in urban habitats, scientists must engage residents (Pickett 2003). Involvement of residents is essential, since we often sample on their properties – and much taxonomic diversity lies in backyards, particularly among small species that have large influences on nutrient cycling (Wilson 1987; Grove 1995; Kim and Byrne 2006). For instance, to look at how lawn care might affect soil arthropod communities involves cooperation from the residents who are asked to fill out questionnaires and allow sampling on their lawn. Methods have to be modified in residential areas, because traditional ecological sampling techniques are sometimes destructive of the habitat (e.g., removal of large quantities of soil) or involve use of potentially harmful substances. Use of water instead of propylene glycol or ethanol as collecting fluid in pitfall traps (McIntyre et al. 2001) requires shorter sampling periods, or

more frequent visits to the sampling sites. Proper assessment of these groups (arthropods, fungi, nematodes, etc) requires taxonomic knowledge or the availability of taxonomic services, both of which are declining (Kim and Byrne 2006).

In our experience, most citizens have been open and willing to help when they can provide valuable data for biodiversity monitoring studies. Such cooperative efforts are successful if good, long-term working relationships with homeowners are maintained, clear and simple instructions are provided about data collection, and when necessary, expert help is provided about their environmental concerns.

In spite of the difficulties of conducting experimental research in cities, there are successes. Cook et al. (2004) illustrate a compelling experiment on landscaping practices and management conducted with the cooperation of a large land owner and local residents. Felson and Pickett (2005) suggest that a closer partnership between scientists and urban designers can help solve this problem more generally. A close collaboration provides the opportunity for rigorous experimentation while creating esthetic and functional urban spaces at the same time. Such spaces are installed for the long term. Thus, the experiments will provide an invaluable tool for monitoring, and reveal urban ecological patterns and processes, including those related to biodiversity.

We must be ready to exploit *natural experiments* such as accidental introductions of new species or major urban development or revitalization projects (Pickett et al. 2007). A project in west Baltimore illustrates many of the characteristics of these experiments. This project combines neighborhood revitalization, greening, removal of unnecessary asphalt, and installing best-management practices for storm water control (Pickett et al. 2007). The interventions involve the government agencies, local communities, and can be used by researchers to improve general scientific knowledge. Importantly, the project also serves to inform local communities about their environment and the way it is changing and is useful in local schools as well.

6.5 Questions and Research Directions on Urban Biodiversity

Throughout the above discussion, we have identified questions and data gaps motivating future research on urban biodiversity. Because the field of urban ecological studies and its integration with social, economic, and physical sciences is evolving so rapidly, we highlight research opportunities in this section.

Variation in biodiversity within urban areas is often associated with such straightforward social and demographic factors as human population density, building density, and the amount of impervious surface (DeGraaf and Wentworth 1986; Munyenyembe et al. 1989; Blair 1996; Clergeau et al. 1998; Fernández-Juricic and Jokimaki 2001). Studies of urbanization gradients have successfully employed such measures to describe urban patterns of diversity for a variety of taxa (Blair 1999; Germaine and Wakeling 2001; Pickett et al. 2001; McKinney 2002). Even when combined to generate multivariate indices of urbanization, however,

these studies often leave much of the variation in diversity across the urban matrix unexplained. This approach does not identify the full richness of social and economic forces driving changes in biodiversity over time.

New approaches are necessary in order to understand the complex human social, cultural, and economic forces underlying changing patterns of biodiversity. Standard human socioeconomic data may identify some of the human activities that influence biological communities (Grove and Burch 1997; Grove et al. 2002). In residential areas, household income may strongly influence the capacity for homeowners to construct diverse plant communities in their neighborhoods (Schmid 1975; Whitney and Adams 1980; Hope et al. 2003; Kinzig et al. 2005). Beyond access to resources, however, differences in lifestyle or ethnicity are associated with differences in landscape preferences, leading to different patterns of biodiversity (Kaplan and Talbot 1988; Fraser and Kenney 2000; Kinzig et al. 2005; Grove et al. 2006b). In Phoenix, AZ, the number of bird species in small urban parks is strongly correlated with the socioeconomic status of surrounding neighborhoods containing higher bird diversity in wealthier neighborhoods (Kinzig et al. 2005). In Baltimore, MD, the nature and cover of vegetation was determined by population density, aggregated neighborhood wealth, or lifestyle indices, depending on whether vegetation in public rights of way, the neglected riparian commons, or private property was examined (Grove et al. 2006b).

Many research questions remain: What mechanisms mediate the relationship between human socioeconomic factors and patterns of species diversity in specific situations? To what extent are these patterns dependent on the aggregate effects of individual human behaviors such as feeding birds or gardening vs. government level decision making such as zoning or redevelopment plans (Kinzig et al. 2005)? Does education about the environment alter decision-making that affects biodiversity such as landscaping decisions (Berkowitz et al. 2003)? What is the relationship of urban human population to the plants and animals that occupy nonurban areas? For most people in the inner city, animals mean pests, including cockroaches, rats, and ants, and perhaps less-than-glamorous species such as house sparrows and pigeons. What kinds of biological diversity do humans value? The answers to these questions are essential to predicting the future outcome of urban biodiversity and to motivating efforts to conserve biodiversity in urban areas – the setting in which the majority of humans now live and work.

The question of urban sustainability looms large as more of humanity finds itself in cities and complex urban agglomerations. In this context, one exciting question is the characteristics of organisms that will successfully maintain populations in a highly modified environment (Chace and Walsh 2006). Genetics, physiology, and behavior are key components (e.g., Harris and Trewheella 1988; Wandeler et al. 2003). However, it will also be necessary to monitor and understand the causes and consequences of biodiversity change in evolving urban areas, because the changes are often nonlinear, and may appear with considerable time-lag (Hansen et al. 2005). The traditional view that urbanization necessarily and uniformly leads to an impoverished flora and fauna must be revised, as the patterns appear to be more complex (Pickett et al. 2001, 2008; Schwartz et al. 2001, 2002). The opposite view

that urban flora and fauna can be enriched due to the presence of many nonnatives is also simplistic (Marzluff and Rodewald 2008).

Biodiversity changes are not only due to direct human influences but also due to the interactions within the biotic community itself. Are the biological forces shaping community structure altered in cities? How did the disappearance from urban habitats of predators, such as wolves, foxes, or cougars, affect the dynamics of prey populations, such as squirrels, rabbits, or deer? How does reemergence of some of these species (Gompper 2002; Morey et al 2007) modify urban trophic structure? Are urban soil food webs highly modified, and do these differences along with altered abiotic conditions affect decomposition processes (Carreiro et al. 1999; Walton et al. 2006)?

What is the effect of human control on prey populations? Are there effective controls possible for emerging outbreaks such as that of white-tailed deer in the eastern US Megalopolis? What are the patterns of herbivory in plant assemblages characterized by introduced, ornamental plants?

The understanding of the urban biota that would emerge from broadly answering the questions posed so far can be considered a prolog to this question about the feedback between biota and the remaining components of urban ecosystems: Do altered patterns of urban biodiversity lead to altered ecosystem functions? For example, how do biomass and primary productivity of lawns compare to natural grasslands? How does kind and level of lawn maintenance affect this function? How do bird feeders change feeding patterns and survival of birds? Are ecosystem functions that yield ecosystem services in urban areas in fact sustainable? Answering this question will require ecologists to understand better the functioning of the novel habitats constructed in urban areas. Most ecological research in cities has taken place in patches that are analogs of the natural or wild habitats' ecologists who have been investigating outside cities for a hundred years. Under what conditions do natives out-compete exotics and vice versa? Do exotic species alter interactions among native species such as predators and their prey, parasites and their hosts, and plants with their pollinators? Many birds and some insects have been observed feeding on nonnative plant species (Reichard et al. 2001; Shapiro 2002). To what extent do these novel food sources and host plants provide usable habitat for native animal species? These questions remain largely unaddressed in the literature.

If humans around the world desire similar features from their environment, then urban areas may converge with one another in their species composition (McKinney and Lockwood 1999; McKinney 2002; Pouyat et al. 2006). This might happen both because humans actively transport the species between continents and because similar characteristics of urban environments select for similar kinds of species – that is, species that can adapt to the presence of humans, their built structures, and the high heterogeneity of urban environments. In addition, urban heat islands, soil moisture conditions, and calcium enrichment from concrete and imported limestone and marble contribute to environmental homogenization (Pouyat et al 2010). So far, there have been few empirical tests of the global homogenization or global convergence hypothesis (but see Blair 2001).

An alternative, not mutually exclusive, possibility is that urbanization could generate new biological diversity through evolutionary responses of species to the novel selection pressures found in cities (Slabbekoorn and Peet 2003). Likewise, different habitats found in different cities might generate divergent selection pressures, resulting in the evolution of new, genetically distinct populations from city to city. House finches residing in highly urbanized settings have shown rapid morphological evolution, forming genetically distinct populations (Badyaev et al. 2000). Tamarisk, an exotic, invasive species in the US, has evolved genetically distinct and novel variants via hybridization, variants that only occur in the US (Gaskin and Schaal 2002). Furthermore, this exotic apparently serves as nesting habitat for a native endangered bird species, the Southwest Willow Flycatcher (Ohmart et al. 1991; Sogge et al. 1997). Introduced populations of gray squirrels in Europe appear to differ behaviorally and perhaps genetically from their North American ancestors (Parker and Nilon 2008). Continuing biodiversity research in urban areas will surely reveal more cases of evolutionary responses to novel habitats created by human actions.

Recently Swan et al. (2010) proposed metacommunity theory as a useful tool to explain patterns of diversity in urban areas. This concept integrates local (environmental factors, species interactions) and regional (species pool, dispersal) factors as forces shaping plant and animal communities. By combining physical, biological, and socioeconomic processes at various scales, the urban metacommunity concept can generate testable hypotheses leading to a better understanding of urban biodiversity.

6.6 Conclusions

Urban biodiversity is at once both an old subject and one that is exploring new frontiers, encompassing the urban fringe and the sometimes rapidly changing patches within old urban cores. Urban areas have been shown to be surprisingly speciose habitats, with combinations of natives and exotics, generalist invaders adapted to human activities, and rapidly evolving specialists on novel and stressful habitats. Foundation studies of urban biota are beginning to be complemented by molecular genetic research, behavioral studies, landscape and dispersal research, and community interaction research. Increasingly, these disciplinary-based endeavors are being placed in an integrative framework that attempts to capture the dynamism of an urban matrix driven by complex feedbacks between ecological, social, economic, and physical processes. The question of human geography and its interaction with biodiversity is core to this rapidly expanding field of urban ecosystem research.

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