Robert A. McCleery
Christopher E. Moorman
M. Nils Peterson *Editors*

Urban Wildlife Conservation

Theory and Practice

Springer

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Editors
Robert A. McCleery
Department of Wildlife Ecology
& Conservation
University of Florida
Gainesville
Florida
USA

Christopher E. Moorman
Department of Forestry and Environmental
Resources
North Carolina State University
Raleigh
North Carolina
USA

M. Nils Peterson
Department of Forestry and Environmental
Resources
North Carolina State University
Raleigh
North Carolina
USA

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Contributors

Lowell W. Adams Department of Environmental Science and Technology, University of Maryland, College Park, MD, USA

Erin E. Boydston U.S. Geological Survey, Western Ecological Research Center, Thousand Oaks, CA, USA

Justin L. Brown Santa Monica Mountains, National Recreation Area, Thousand Oaks, CA, USA

Michael E. Dorcas Department of Biology, Davidson College, Davidson, NC, USA

David Drake Department of Forest and Wildlife Ecology, University of Wisconsin-Madison, Madison, WI, USA

Stanley D. Gehrt School of Environment and Natural Resources, The Ohio State University, Columbus, OH, USA

Nia Haynes Department Wildlife Ecology and Conservation, University of Florida, Gainesville, FL, USA

Dustin L. Herrmann Department of Plant Sciences, University of California, Davis, Davis, CA, USA

George R. Hess Department of Forestry & Environmental Resources, North Carolina State University, Raleigh, NC, USA

Mark Hostetler Department of Wildlife Ecology & Conservation, Institute of Food & Agricultural Sciences, University of Florida, Gainesville, FL, USA

Susan K. Jacobson Department Wildlife Ecology and Conservation, University of Florida, Gainesville, FL, USA

Anna L. Johnson Department of Geography & Environmental Systems, University of Maryland, Baltimore County, Baltimore, MD, USA

Courtney L. Larson Colorado State University, Fort Collins, USA

Robert A. McCleery Department of Wildlife Ecology & Conservation, Institute of Food & Agricultural Sciences, University of Florida, Gainesville, FL, USA

Melissa R. McHale Department of Forestry and Environmental Resources, North Carolina State University, Raleigh, NC, USA

Nancy E. McIntyre Department of Biological Sciences, Texas Tech University, Lubbock, TX, USA

Christopher E. Moorman Fisheries, Wildlife, and Conservation Biology Program, Department of Forestry and Environmental Resources, North Carolina State University, Raleigh, NC, USA

Joanne G. Moriarty Santa Monica Mountains National Recreation Area, National Park Service, Thousand Oaks, CA, USA

Charles Nilon Department of Fisheries and Wildlife Sciences, University of Missouri, Columbia, MO, USA

Sarah R. Partan School of Cognitive Science, Hampshire College, Amherst, MA, USA

M. Nils Peterson Fisheries, Wildlife, and Conservation Biology Program, North Carolina State University, Raleigh, NC, USA

Steven J. Price Department of Forestry, University of Kentucky, Lexington, KY, USA

Sarah Reed Wildlife Conservation Society North America Program, Bozeman, MT, USA

Seth P. D. Riley Santa Monica Mountains National Recreation Area, National Park Service, Thousand Oaks, CA, USA

Amanda D. Rodewald Cornell Lab of Ornithology and Department of Natural Resources, Cornell University, Ithaca, NY, USA

Amy M. Ryan Neuroscience and Behavior Program, University of Massachusetts Amherst, Amherst, MA, USA

Ryo Sakurai Department Wildlife Ecology and Conservation, University of Florida, Gainesville, FL, USA

Catherine M. Schoonmaker Santa Monica Mountains, National Recreation Area, Thousand Oaks, CA, USA

Kirsten Schwarz Department of Biological Sciences, Northern Kentucky University, Highland Heights, KY, USA

Laurel E. K. Serieys Dept. of Ecology and Evolutionary Biology, University of California, Los Angeles, South, Los Angeles, CA, USA

Jeff A. Sikich Santa Monica Mountains, National Recreation Area, Thousand Oaks, CA, USA

Joel W. Snodgrass Department of Biological Sciences, Towson University, Towson, MD, USA

Michael W. Strohbach Department of Environmental Conservation, University of Massachusetts—Amherst, Amherst, USA

Department of Geography, Humboldt University, Berlin, Germany

Christopher M. Swan Department of Geography & Environmental Systems, University of Maryland, Baltimore County, Baltimore, MD, USA

Janette Thompson Iowa State University, Ames, USA

Dara M. Wald Department Wildlife Ecology and Conservation, University of Florida, Gainesville, FL, USA

Paige S. Warren Department of Environmental Conservation, University of Massachusetts—Amherst, Amherst, USA

About the Editors

Robert A. McCleery is an Assistant Professor in the department of Wildlife Ecology and Conservation Biology at the University of Florida. His research focuses understand how wildlife responds to anthropogenic changes to their environment and finding ways to maintain wildlife communities and populations that foster healthy ecosystems. Much of Bob's work has been centered on mammalian conservation in urbanizing and agricultural landscapes. Bob received his B.S. from Cornell University and M.S. and Ph.D. degrees from Texas A&M University. Additional information about his research, teaching, and outreach activities can be viewed on his web page: http://www.wec.ufl.edu/faculty/mccleery/.

Department of Wildlife Ecology & Conservation, University of Florida, Gainesville, FL, USA

Christopher E. Moorman is Professor and Coordinator of the Fisheries, Wildlife, and Conservation Biology Program at North Carolina State University (NCSU). His research focuses on global change and wildlife with emphasis on understanding the effects of human-induced landscape change on wildlife habitat. Chris developed the course Urban Wildlife Management at NCSU, published a 4-part extension publication series on backyard and urban wildlife management, and led collaborative efforts to create the Going Native website (http://ncsu.edu/goingnative/), an interactive resource for those interested in landscaping for wildlife with native plants. Chris received his B.S. and M.S. degrees from the University of Georgia, and his Ph.D. from Clemson University. Additional information about his research, teaching, and outreach activities can be viewed on his web page: http://www4.ncsu.edu/~moorman/.

North Carolina State University, Raleigh, NC, USA

M. Nils Peterson is an Associate Professor of Fisheries, Wildlife, and Conservation Biology at North Carolina State University. His research focuses on unravelling the drivers of environmental behavior, using environmental education, conservation development, environmental conflict, and environmental policy making as natural experiments to test hypotheses. Much of this research is summarized in his recent book The Housing Bomb. Nils received his B.S. and M.S. degrees from Texas A&M University, and his Ph.D. from Michigan State University. Additional information about his research, teaching, and service activities is on his web page: http://www4.ncsu.edu/~mnpeters/index.htm.

North Carolina State University, Raleigh, NC, USA

Chapter 1
Introduction

Robert A. McCleery, Christopher E. Moorman and M. Nils Peterson

More than 50 % of the world's human population lives in urban and suburban areas, and over 67 % of humans will live in cities by 2050 (United Nation Population Division (UNPD) 2012). Although urban areas only account for 2.8 % of the earth's land surface (Millennium Ecosystem Assessment 2005), they exert a dominant influence on regional and global ecological systems through intense resource use and waste production (Collins and Kinzig 2000). Urban centers are characterized by dense concentrations of people, buildings, impermeable surfaces, introduced vegetation, and some wildlife species. The high concentrations of people in urban centers are maintained through imported food, water, and energy, which are turned into sewage and pollution, which are exported to other landscapes (McDonnell and Pickett 1990; Pickett et al. 2001; Adams et al. 2006). At a broader scale, urban landscapes are mosaics of residential, industrial, and commercial buildings interspersed with green areas (Breuste et al. 2008). These green areas, important for wildlife, are in turn constituted by patches of lawn, parks, trails, golf courses, cemeteries, and remnants of native vegetation.

Historically, urban areas were rarely considered potential wildlife habitat and neglected by wildlife ecologists and managers. Wildlife and the ecosystems on which they depend were seen as things that persisted in places away from cities and human

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R. A. McCleery (✉)
Department of Wildlife Ecology & Conservation, Institute of Food & Agricultural Sciences,
University of Florida, 110 Newins-Ziegler Hall, PO Box 110430,
Gainesville, FL 32611-0430, USA
e-mail: ramccleery@ufl.edu

C. E. Moorman
Fisheries, Wildlife, and Conservation Biology Program, Department of Forestry
and Environmental Resources, North Carolina State University,
110 Brooks Avenue, Box 7646, Raleigh, NC 27695, USA
e-mail: chris_moorman@ncsu.edu

M. N. Peterson
Fisheries, Wildlife, and Conservation Biology Program, North Carolina State University,
Raleigh, NC 27695-7646, USA
e-mail: nils_peterson@ncsu.edu

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influences (Worster 1994; Peterson et al. 2007). Wildlife ecologists actively sought out study areas far from civilization in the hope of uncovering facts untainted by human influences. This approach worked well when places without urban influence existed. In today's urbanizing world such an approach is woefully inadequate. Not only do most humans live in urban areas, but the area of urban settlements is growing exponentially and housing development is most rapid in biodiversity hotspots (Liu et al. 2003). Even the shrinking portion of our population living on farms (≈2 % in the USA) is primarily providing resources to those living in urban environments. Urban wildlife conservation responds to this global shift by rejecting the myopic focus on wildlife in natural systems and embracing a focus on wildlife in the urban systems which dominate human land use today.

One major reason wildlife science has increasingly focused on urban environments is the realization that the most pressing wildlife conservation challenges are urban in nature. Sprawl from urban areas remains a primary threat to native species and biodiversity in adjacent agricultural areas and wildlands (McKinney 2002; Pickett et al. 2008). Urban development also removes and fragments habitats, thereby threatening long-term conservation of wildlife species less able to persist in smaller, more isolated habitat patches. Additionally, development of water, communication, and energy infrastructure for cities present some of the fastest growing global threats to wildlife conservation (Chaps. 3, 10, 13–16).

Further, the proximity of humans and wildlife in urban areas leads to more human–wildlife interactions than in any other setting, so, wildlife have more impact on human health, quality of life, education, and esthetics in urban areas than in any other place (McCleery et al. 2012). Higher biodiversity tends to improve the health of humans along with their perceived quality of life (Fuller et al. 2007; Sala et al. 2009). Urbanities show measurable physical and psychological benefits associated with time in urban green space, and these benefits increase with greater species richness in the green-spaces (Fuller et al. 2007). Alternatively, the urban adapted species that people frequently observe and interact with can serve as hosts, vectors, and reservoirs for zoonotic diseases that pose health threats to humans (Chap. 10). The frequent interactions between humans and wildlife in urban areas also creates a need to find safe, humane, and socially acceptable means to reduce property damage from overabundant wildlife populations (Chap. 17). Finally, the proximity of people and wildlife in urban areas provides excellent educational opportunities to foster an understanding of sound ecological principles (Pickett et al. 2008; Stevenson et al. 2013).

1.1 Urban Wildlife Conservation

Urban wildlife conservation examines wildlife in areas dominated or influenced by the built environment, with a focus on the ecology of wildlife and the interactions of wildlife with humans and other features of the urban ecosystem. This field is not confined to metropolitan areas or cities and encompasses the great variety of urban forms and built environments present across the planet. Moreover, urban wildlife conservation can extend far beyond the built environment, following the transportation networks, water sources, utilities, food products, and pollutants that feed into and emanate from urban systems (Chaps. 3, 10, 12, 15, 16). Scientists often use the term "urban" and assume their audience knows the meaning (McDonald and Pickett 1990; McIntyre et al. 2000; Marzluff et al. 2001). This creates a problem because most of the places where wildlife scientists work have been modified by humans (Ellis and Ramankutty 2008), and standard designations for human altered landscapes do not exist. Further, names that have been given to human-altered landscapes (urban, golf course, urban park, suburban, exurban, rural, city core, city center, urban-wildlands interface, etc.) rarely specify how the landscape has been altered. As urban wildlife conservation research expands, defining the sub-discipline and documenting the extent of urbanization associated with study sites are becoming critical.

Due to the breadth of research associated with urban influences, it is important not to simply describe any area developed by humans as urban but to describe the extent of human development and alteration in and around each study site. Marzluff et al. (2001) suggested using development, buildings, and human densities to create distinct categories (urban, suburban, rural and wildlands) along an urban–rural gradient. A more precise approach requires quantifying the features of the urban environment that are hypothesized to influence wildlife on a particular study site (e.g., percent impervious surface; Arnold and Gibbons 1996; McKinney 2002). Land cover type, age since conversion, road density, traffic patterns, and human population density can all be quantified to help explain the distribution, demography, or behaviors of urban wildlife (McIntyre et al. 2000; McCleery et al. 2012)

Urban systems are fundamentally unique because human activities drive their spatial patterns, ecological processes, and dynamics over time (Warren et al. 2010). Because of these interacting factors, urban wildlife conservation is more complex and interesting than traditional wildlife conservation. For instance, the prevalence of white-tailed deer (*Odocoileus virginianus*) in urban systems is connected to the rise of low density suburbs during the process of urban sprawl, which in turn can be tied to federal subsidies for highways, federal home loan insurance, and a legacy of racism leading to "white flight" from racially diverse inner cities (Soule 2006; Gonzalez 2009), in addition to climate and all the other variables a traditional wildlife ecologist might consider. In short, urban wildlife conservation requires a difficult balancing act where both ecological and social dynamics relevant to a particular wildlife related question must be considered simultaneously (Chap. 3). Accordingly, as a field, urban wildlife conservation has emerged as an interdisciplinary effort including ecological and social research, urban planning, outreach, education, and management. Although urban wildlife conservation is interdisciplinary in nature, the ultimate goal of this scientific endeavor is to understand, manage, and create built environments conducive to wildlife and people alike.

1.2 The Rise of Urban Systems

The urban environment is new. It is new to humans and in evolutionary terms, it is new to wildlife species. It was not until 5000–6000 BC with the advent of the ox-drawn plow, irrigation and the domestication of new plants that the first small cities arose in fertile flood plains and valleys (Davis 1955). These first cities were small, encompassing only several square kilometers with less than 50,000 inhabitants (Davis 1955). Early cities were limited in size by the productivity of the lands surrounding them (Mumford 1956). Around 1000 BC with improvements to transportation (roads, chariots, oxcarts), communication and commerce, cities expanded their access to agricultural products and resources. As a result, cities grew in size (Greek and Roman cities >100,000) and increasingly specialized in trade and industry (Davis 1955; Mumford 1956).

Rapid expansion and growth in urban areas did not begin until the last several centuries. At the turn of the nineteenth century, only 50 cities had more than 100,000 people, and only 10 % of the planet's human population lived in urban areas (Davis 1955; Mumford 1956; Grimm et al. 2008). Over the last 200 years, urban areas have become centers of commerce, transportation, culture, government and social interactions. As of 2011, over 52 % of the planet's roughly 7 billion people lived in urban areas, and there were over 1400 cities with populations over 1,000,000 people (United Nation Population Division 2012). The growth of modern cities has been driven by the industrial revolution, population growth, and the green revolution (Davis 1965). As agriculture was transformed with fertilizer, controlled irrigation, improved grain varieties, and mechanization, a smaller portion of the population was needed for food production, helping to drive migration to urban centers with economies focused on industry and trade. The growth of our urban areas is expected to continue. By 2050, 67 % of the world's inhabitants are projected to live in urban areas (United Nation Population Division (UNPD) 2012). From 2011 to 2050, we are projected to add 2.3 billion people to the planet and 2.6 billion people to urban areas from births and immigration (United Nation Population Division (UNPD) 2012). Along with the growth in human population will come a growth in the size of cities, all of which will need a continuous supply of resources and ecosystem services (Grimm et al. 2008). Currently, urban areas only account for about 2.8 % of Earth's total land area (Millennium Ecosystem Assessment 2005), but due to the intensity of resource use in urban areas, urban systems influence regional and global ecological systems and processes (Collins and Kinzig 2000). Cities are not isolated island systems; rather, they are hot spots that propel environmental change on multiple scales (Grimm et al. 2008).

In general, the environmental changes associated with urbanization have not been favorable for wildlife. In fact, vertebrate species richness decreases with urbanization, and urbanization has been a primary cause of extinction and local extirpation of native fauna (Czech et al. 2000; McKinney 2002, 2008). Yet, it is wrong to see urban areas as ecological deserts. Many cities were built on some of the planet's most productive lands (Ullman 1941) and have ample resources that allow native species to persist, adapt, and thrive (McKinney 2002, Chaps. 8, 9). In fact, urban areas shelter and enhance the populations of some endangered species and are even home to newly discovered species (McCleery 2010; Newman et al. 2012).

1.3 Urban Footprint

As the human population becomes more urbanized, we have not lessened our dependence on nature to supply us with food and natural resources (Bolund and Hunhammar 1999). Cities must import the energy and resources necessary to sustain human life. However, advances in technology and transportation allow modern cities to meet their needs by accessing resources from all corners of the planet (Luck et al. 2001). This continuous influx and consumption of resources have created a suite of unique ecological conditions (Chap. 4). Cities have distinctive vegetative communities (Chap. 5), climates, soils, hydrology and disturbance regimes (Chap. 4) that drive ecological process and shape wildlife communities within them (Chap. 7). On regional and global scales, cities alter environments and land uses through their consumption of resources. For example, 60 % of residential water use and > 75 % of all industrial wood products are consumed by cities (Grimm et al. 2008).

Some ecologists and economists have tried to conceptualize the influence of cities on surrounding lands and resources by calculating an ecological footprint, or area of productive land necessary to supply a population (Rees and Wackermagel 1996). Early calculations suggested that 5.1 ha of productive land were necessary to meet the resource needs of every American (Rees and Wackermagel 1996). Not surprisingly, the concept of an urban ecological footprint was criticized for assuming the area necessary to supply different resources to each city was equivalent (Luck et al. 2001). Still, even as an imperfect concept, the urban footprint clearly illustrates the influence that cities can exert over areas tens to hundreds of times greater than their physical footprint (Rees and Wackermagel 1996; Grimm et al. 2008). More recently, revised and nuanced ecological footprint models account for spatial heterogeneity in resource access and location, and differences in the size, and demands of cities (Luck et al. 2001). These models suggest the catchment area needed to supply water and food to New York City is half of what is needed for Los Angeles (Luck et al. 2001). New York City is home to twice as many people, but Los Angeles covers more land area, is surrounded by arid and less productive lands, and must access resources from more remote areas.

Cities also influence ecological processes far beyond their boundaries through the exportation of their waste and byproducts. Cities are a major source of trace metals, CO_2, NO_2, O_3, SO_2, and particulate air pollution (Akimoto 2005; Grimm et al. 2008). Seventy-eight percent of all carbon admissions come from cities, and the concentrated release of heat and carbon from fossil fuel in urban environments has increased temperatures within cities and altered global climate patterns (Bulkeley and Betsil 2003). Thus, even climate change related threats to polar bears (*Ursus maritimus*) are driven in part by urban systems. Increased urban temperatures may also help to create smog and disperse pollutants away from cities (Grimm et al. 2008). Furthermore, the impervious surfaces created in urban areas cause drastic changes in the hydrology and quality of water adjacent to and downstream from cities. Impervious urban surfaces have impacted streams by increasing maximum flows, water temperatures, nitrogen, phosphorus, heavy metals, pesticides, and eutrophication (Paul and Meyer 2008, Chaps. 4, 16).

In summary, it is wrong to view cities as isolated self-contained systems. Cities alter ecosystem functions and process far beyond their borders, and thus any discussion of urban ecology and urban wildlife conservation should not be bound to the city limits. Whereas the growth and expansion of cities has directly expatriated some wildlife and allowed other species adapt to the environments, the rapidly expanding footprint has increased the indirect effects on wildlife through habitat conversion to agriculture, altered hydrology, and water projects, increased energy production, and climate change.

1.4 Changing Relationships with the Land

The relatively new ability of humans to create a surplus of food and transport it efficiently to centralized locations (cities) has inherently changed humanity's relationship with nature (Wirth 1938; Davis 1955). The prominent wildlife ecologist and philosopher Aldo Leopold believed the disconnection between people and their food source was a moral problem leading to abuse of the land (Leopold 1966). As urban dwellers have become disconnected from their food source and agriculture, they have more generally become removed from wildlife (Peterson et al. 2010), with most urbanites inhabiting regions of their city with low biodiversity (Turner et al. 2004; Miller 2005). A byproduct of the separation between humans and nature is a diminished understanding and appreciation for biodiversity conservation (Hough 1995). It is feared that as people become increasingly ignorant of nature and devalue their experiences in the natural world, they will be less inclined to invest in and conserve natural areas and their associated biodiversity (Miller 2005). Furthermore, as generations of urban dwellers become accustomed to degraded, low diversity environments they will lower their expectation of access to quality natural settings (Miller 2005).

The separation between people and the outdoors also has consequences for human health and well-being (Rohde and Kendle 1994), development of children (Kellert 2002), and the identity of communities (Horwitz et al. 2001). Scientists recently have begun to quantify how contact with nature benefits people. Research shows spending time in the outdoors leads to a positive emotional state and reduced stress levels (Ulrich et al. 1991). More specifically, increased biodiversity in all environments improves the health of humans along with their perceived quality of life (Fuller et al. 2007; Sala et al. 2009). Even urbanities experiencing nature in urban green-spaces have shown measurable physical and psychological benefits, and these benefits increased with increased species richness in the green-spaces (Fuller et al. 2007). The benefits of contact with nature begin at an early age. For example, a child's intellectual and emotional capacities are enhanced by his or her interaction with nature in a familiar setting (Kellert 2002; Miller 2005).

1.5 Potential for the Urban Areas

The creation of cities has helped to separate humans from nature, but paradoxically cities hold enormous potential to re-engage humans with the natural world. In fact, biodiversity conservation may depend on the reconnection of humans to nature in urban areas (Turner et al. 2004). Human appreciation and understanding of biodiversity increases when diversity can be found close to home (Hough 1995).

Therefore, the logical places to foster conservation efforts are within cities. This task might not be as daunting as it sounds, as urbanities prefer urban environments with prominent natural elements (Herzog 1989). Furthermore, people respond to all kinds of biodiversity, and both native and nonnative species can help counter the deficit of nature in cities (Louv 2005; Miller 2005). Thus, it is not necessary to restore pristine environments within urban centers. We simply need to create pockets of diversity and natural features within cities that that are safe and easily accessed by urbanities and their children (Miller 2005).

One of the most direct and rewarding ways for people to experience nature is to observe and interact with wildlife. As demonstrated in this book, there is a growing body of expertise on how to attract and retain wildlife in urban areas on multiple scales. Actions taken in backyards, neighborhoods, and throughout cities can create environments that allow urbanities to experience wildlife in positive manner, thus enhancing their interactions with nature (Chaps. 12–14). The ability to attract and retain wildlife in urban areas also provides an ideal opportunity for outreach, education, and public involvement to help develop an understanding and appreciation of ecological relationships and conservation goals (Pickett et al. 2008; Stevenson et al. 2013). Ironically, it is in urban settings where we have the greatest potential to reconnect humans to wildlife and nature; doing so concurrently promotes natural resource conservation and enhances the health and well-being of urban inhabitants.

1.6 Overview of Book

In this book we hope to showcase the breadth of the expanding field of urban wildlife conservation. We also attempt to draw attention to research and knowledge that has developed over the last several decades. Recent areas of focus for researchers and practitioners include:

1. Understanding the influence of built environment and human activity on wildlife behaviors, population dynamics, physiology, and interspecies interactions
2. Exploring the linkages among social status, economics, and politics and wildlife habitats at various scales
3. Planning and developing built environments that accommodate wildlife
4. Understanding interactions between wildlife and humans
5. Informing transformative management of wildlife damage

We have attempted to summarize over three decades of literature on urban wildlife into a cohesive narrative of the subject. We focused the book on understanding urban wildlife communities and properly applying scientific knowledge to the creation of urban environments that are beneficial to humans and wildlife alike. By synthesizing new information on these topics and placing them into an interdisciplinary context, we aim to provide a foundation for the future growth and understanding of urban wildlife conservation. The book is divided into four sections. In the Introductory section (Chaps. 1, 2), we define urban wildlife science and explore the history of urbanization and the field of urban wildlife conservation. In the Urban System section, we explore the complex ecological and social drivers that shape the urban environments (Chaps. 3–5). In the Wildlife Response to Urbanization section, we describe the patterns of wildlife response to an urbanizing world and examine the mechanisms creating these patterns (Chaps. 6–9). We conclude the book with the Living Together section that introduces ways to create physical and social environments that are mutually beneficial for humans and wildlife (Chaps. 10–17).

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Chapter 2
History of Urban Wildlife Conservation

Lowell W. Adams

Keywords Social need · Ecological landscapes · Metropolitan Open Space System · Urban Wildlife Working Group · Urban Wildlife Committee · Urban Wildlife Research Center

2.1 In the Beginning

Urban wildlife management is rooted in game management. According to Leopold (1933), game management was first practiced in Asia by Kublai Khan during the latter half of the thirteenth century (Fig. 2.1). At that time, game animals could not be taken between March and October. Such practice found its way to Europe where a long history developed of setting hunting seasons and bag limits to manage game species. *The Master of Game* is considered the oldest English book on hunting, written by Edward of Norwich, Second Duke of York, between 1406 and 1413 (Baillie-Grohman and Baillie-Grohman 2005) (Fig. 2.2). The practice of setting hunting seasons and bag limits to manage game species was transferred to North America with European settlement of the continent. For example, Rhode Island closed the hunting season for white-tailed deer (*Odocoileus virginianus*) from May to November in 1639 and Iowa established a bag limit of 25 greater prairie chickens (*Tympanuchus cupido*) per day in 1878 (Leopold 1933 as cited in Bolen and Robinson 2003).

Game management as a science in the United States began in the 1930s, led by the publication of Aldo Leopold's book *Game Management* in 1933 (Leopold 1933). The Cooperative Wildlife Research Unit program began in 1935. In 1936, The Wildlife Society was formed and the first North America Wildlife Conference (now North American Wildlife and Natural Resources Conference) was held. In 1937, the Federal Aid in Wildlife Restoration Act was enacted. That legislation is widely known as the Pittman–Robertson Act (or P–R Act), and is one of the most important wildlife acts in the USA.

L. W. Adams (✉)
Department of Environmental Science and Technology,
University of Maryland, College Park, MD 20742, USA
e-mail: Ladams4@umd.edu

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Fig. 2.1 Game management was first practiced in Asia by Kublai Khan during the latter half of the thirteenth century. Kublai Khan prohibited the taking of game animals between March and October. Later the practice of game management was adopted in Europe and from there it spread to North America

During the early years of wildlife management in North America, far-sighted wildlife biologists recognized the importance of urban wildlife. For example, in his classic text, Aldo Leopold stated "A pair of wood thrushes (*Hylocichla mustelina*) is more valuable to a village than a Saturday evening band concert, and costs less" (Leopold 1933, p. 404). Rudolf Bennitt, first president of The Wildlife Society, summarized the 1946 North American Wildlife Conference and stated: "I still look forward to the day when we shall hear men discuss the management of songbirds, wildflowers, and the biota of a city" (Bennitt 1946, p. 517).

The 1960s saw greater focus on urban areas with regard to wildlife conservation and management. In 1966, Raymond Dasmann, another prominent wildlife biologist soon to be president of The Wildlife Society, spoke of "old conservation," concerned mainly with quantity of natural resources, and "new conservation," dealing principally with clean air and water, open space, outdoor recreation, and quality of the human environment, particularly the urban environment, where most people live (Dasmann 1966). He pointed out that generations of humans were growing up in cities with no roots in the land and little experience in the natural world. Dasmann believed the wildlife profession was too closely identified with game animals and hunters, and was too narrow-minded. He believed that more wildlife biologists should focus their efforts on the metropolitan environment.

In 1968, the US Bureau of Sport Fisheries and Wildlife (now US Fish and Wildlife Service) sponsored a national conference on "Man and Nature in the City." Then, the Bureau Director Dr. John Gottschalk stated: "If our Bureau were to focus, as we have in the past, on the wide open spaces and neglect the people in the city,

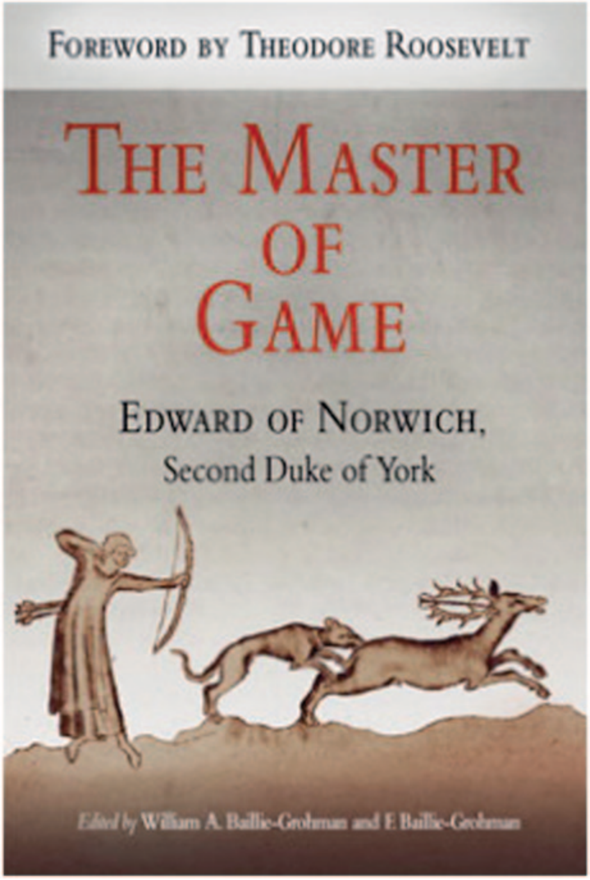

Fig. 2.2 *The Master of Game* is considered the oldest English book on hunting, written by Edward of Norwich, 2nd Duke of York, between 1406 and 1413. In addition to discussing hunting practices, the book includes chapters on the nature of various animals as known at the time

I believe it would soon find itself in a very questionable orientation with society" (Bureau of Sport Fisheries and Wildlife 1968, p. viii).

2.2 Growth of Urban Wildlife Conservation

This brings us to the late 1960s and 1970s when the discipline of urban wildlife ecology, conservation, and management grew more rapidly. I begin this section focusing first on the people and institutions of the USA and ending with international programs and activities.

2.2.1 Urban Wildlife Working Group of The Wildlife Society

In the late 1960s, Al Geis, Bob Dorney, and other members of The Wildlife Society proposed that a committee of that society be formed to focus on wildlife and urban areas. The committee was established as the Urban Affairs and Regional Planning Committee. That committee reached out to landscape architects and planners with the realization that those professionals played important roles in urban areas. In 1975, wildlife biologists were invited to convene a session at the American Institute of Planners Meeting. The following year, planners and landscape architects were invited to convene a session on planning and design at the North America Wildlife and Natural Resources Conference. The committee was renamed the Urban Wildlife Committee in 1982.

The Urban Wildlife Committee was quite active. In 1983, it prepared a policy statement on urban wildlife that was adopted by Council of The Wildlife Society on 11 October of that year. The statement highlighted wildlife as an important component of the urban environment (The Wildlife Society 2012).

The committee conducted two surveys of urban programs. The first was a 1983 survey of state conservation agencies in the USA. In that survey, six agencies reported the existence of designated urban wildlife programs (Lyons and Leedy 1984). The principal functions of those programs were extension, public education, and management. Only three states reported that research was a part of their program activities. The second survey was conducted in 1985 and focused on North American colleges and universities (Adams et al. 1987). About 2 % of wildlife research budgets were devoted to urban wildlife studies in 1983–1984. Few schools (9 %) offered specific courses in urban wildlife, but most (78 %) included the topic in other wildlife courses.

The committee prepared and published a report entitled "Guidelines for Implementing Urban Wildlife Programs Under State Conservation Agency Administration" (Tylka et al. 1987). The report recommended four main elements for a well-rounded urban wildlife program: (a) inventory and research, (b) planning and management, (c) public information, education, and extension services, and (d) urban habitat acquisition, development, preservation, restoration, and conservation. The committee also assisted the American Society of Landscape Architects in developing a policy statement on wildlife and wildlife habitat in 1988. The policy statement reads in part "Landscape architecture, allied design professions, and wildlife management apply similar principles to planning for the beneficial use of the land and support an awareness of and appreciation for wildlife, wildlife habitat, and their value to the planet. The Society therefore urges the identification and application of planning and design principles that promote the enhancement, protection and management of landscapes that support wildlife" (American Society of Landscape Architects 2013, p. 1). In 1999, the Urban Wildlife Committee evolved into the Urban Wildlife Working Group of The Wildlife Society. The working group has been active in sponsoring workshops and sessions at annual meetings of The Wildlife Society (Fig. 2.3). And, following the

Fig. 2.3 In recent years, the Urban Wildlife Working Group of The Wildlife Society has sponsored workshops and field trips at the Society's annual conference. Shown here are participants who took part in an all-day workshop in Tucson, Arizona, 22 September 2007. The workshop included a desert walk at the Arthur Pack Regional Park and Tucson Audubon's Mason Center. Both areas protect saguaro-ironwood desert habitat of northwest Tucson

fourth urban wildlife symposium in Tucson, Arizona, in 1999, it assumed a leadership role in continuing the symposium series initiated by the National Institute for Urban Wildlife in 1986 (Sect. 2.2.3).

2.2.2 Urban Wildlife Research Program of the US Fish and Wildlife Service

The US Fish and Wildlife Service officially established an urban wildlife research program in June 1972 (Geis 1981). The program focused on birds and was headed by Dr. Aelred D. Geis of the Service's Patuxent Wildlife Research Center central campus in Laurel, Maryland (Fig. 2.4). Early in his program, Dr. Geis studied bird-habitat associations in relation to development of the new town of Columbia, Maryland, and documented bird community changes as development advanced (Geis 1974a, b, 1976). Geis noted that farmland and field species, such as northern bobwhite (*Colinus virginianus*) and eastern meadowlark (*Sturnella magna*), declined, and other species, such as northern mockingbird (*Mimus polyglottos*) and song sparrow (*Melospiza melodia*), increased. He also found that building design and

Fig. 2.4 Dr. Aelred Geis of Patuxent Wildlife Research Center was head of the US Fish and Wildlife Service's urban wildlife research program that was established in 1972. Dr. Geis's early work focused on bird-habitat associations. Later he conducted research on bird feeding and published a report on the subject that received high public demand and was distributed widely

quality of construction affected density of house sparrows (*Passer domesticus*) and European starlings (*Sturnus vulgaris*) and unboxed eaves provided small openings beneath house roofs that these birds used for nesting sites.

Dr. Geis also studied birds in a wooded natural area of Baltimore, Maryland, and a nearby residential area of detached and two-family attached housing (Geis 1980a). He reported the highest density of birds and lowest number of species in the mature residential area and the lowest density of birds but highest number of species in the wooded natural area. This work helped to establish a pattern of bird density and diversity in relation to urban development that is now well accepted in the scientific community (See Chaps. 7 and 8).

Based on his bird-habitat research, Dr. Geis developed planning and management recommendations for urban and urbanizing areas. He argued that trees and shrubs preserved or planted in urban open spaces were valuable for wildlife and that urban open space should be better managed. He believed that too much public open space was simply mowed and could be managed in a more sound ecological way that would provide better wildlife habitat and offer wildlife viewing opportunities to people.

Dr. Geis conducted research on supplemental bird feeding by people, work that focused on seeds birds liked to eat. He found that the small, oil-type sunflower seed and white proso millet were best for use under Maryland conditions. At the time, oil sunflower seeds were not marketed as birdseed. Geis's work created demand by the public and he played a role in convincing the seed industry to make oil sunflower available as birdseed. Geis published his research as a US Fish and Wildlife Service Special Scientific Report, which was distributed widely (Geis 1980b). See Adams (2012) for more detail regarding Dr. Geis's urban research.

Dr. Geis was active in The Wildlife Society and was influential in its establishment of an Urban Affairs and Regional Planning Committee (above). He also was

Fig. 2.5 The core of the Urban Wildlife Research Center for many years. From *left* to *right*, Louise Dove, Lowell Adams, Dan Leedy, Tom Franklin, and Barbara McFalls

a major force behind creation of the Urban Wildlife Research Center (above). During the 1970s and 1980s, particularly, Dr. Geis was a strong advocate in expressing need for the wildlife profession to get more involved in urban areas.

2.2.3 Urban Wildlife Research Center

The Urban Wildlife Research Center was founded in 1973 as a private, nonprofit scientific and educational organization dedicated to wildlife conservation in urban, suburban, and urbanizing areas (Adams 1989) (Fig. 2.5). It was renamed National Institute for Urban Wildlife in 1983 and closed in 1995. Most of the work of the organization resulted in scientific, technical, or popular publications. Examples include an early literature review (Leedy 1979), planning considerations for fish and wildlife (Leedy et al. 1978, 1981; Adams and Dove 1989), proceedings of two national symposia on urban wildlife (Adams and Leedy 1987, 1991), and two educational primers (Adams and Dove 1984; Leedy and Adams 1984). Beginning in 1975, in cooperation with the Urban Affairs and Regional Planning Committee of The Wildlife Society, the Center organized and held annual open exchange meetings in conjunction with the North American Wildlife and Natural Resources Conference. The meetings were designed to provide an opportunity for those interested in urban wildlife to get together and discuss programs, policies, and research and management activities. In 1986, the Institute initiated an urban wildlife symposium series to bring together biologists, landscape architects, planners, and other

Fig. 2.6 The National Wildlife Federation encourages homeowners to think about wildlife needs. In 1973, it developed a Backyard Wildlife Habitat Program (now the Certified Wildlife Habitat Program). Shown here is a front yard in Columbus, Ohio, that has been certified by the Federation. (Photo by Toni Stahl)

professionals working in urban, suburban, and urbanizing areas. Proceedings of the first two and fourth conference were published (Adams and Leedy 1987, 1991; Shaw et al. 2004). The fifth symposium was held in Massachusetts in 2009 and the sixth in Texas in 2011. Those meetings provided an excellent forum for wildlife biologists and others to get together and discuss the art and science of wildlife conservation and management in metropolitan environments.

2.2.4 National Wildlife Federation

The National Wildlife Federation was involved early on with urban wildlife conservation, primarily with initiation of its Backyard Wildlife Habitat Program in 1973 (Tufts 1987) and publication of *Gardening With Wildlife* the following year (National Wildlife Federation 1974). Tufts and Loewer (1995) authored a follow-up to the latter publication. The Backyard Wildlife Habitat Program (now the Certified Wildlife Habitat Program) remains popular. It is designed to educate and motivate citizens to enhance urban wildlife habitat in their own backyards, schoolyards, and other properties, and it certifies habitats that meet established criteria (Fig. 2.6). Some 4700 habitats were certified by 1986 (Tufts 1987). More than 150,000 habitats were certified by May 2012 (National Wildlife Federation 2012). The concept has expanded to other private organizations (The Humane Society of the United States 2012), as well as state wildlife agencies (Penland 1987; Bender 2004).

2.2.5 The State University of New York, Syracuse

In the mid 1970s, Dr. Larry W. VanDruff of the State University of New York, Syracuse, developed the first, or one of the first, graduate courses and programs in the US in urban wildlife ecology (Fig. 2.7). The program focused mostly on birds

Fig. 2.7 Dr. Larry VanDruff of the State University of New York was an early leader in urban wildlife conservation and management in the USA. He helped to train some of the first wildlife biologists who specialized in urban wildlife

and mammals. Dr. VanDruff and his students were active in regional and national wildlife conferences, urban wildlife symposia, and the urban wildlife committee of The Wildlife Society. Dr. VanDruff chaired the urban wildlife committee during a portion of the 1980s. He was awarded the Daniel L. Leedy Urban Wildlife Conservation Award of 1987 by the National Institute for Urban Wildlife for outstanding professional commitment and contributions to the conservation of wildlife and habitat in urban, suburban, and developing areas. Dr. VanDruff's graduate students included Bob Bruleigh, Art Johnsen, Charlie Nilon, and Mike O'Donnell, among others.

2.2.6 The USDA Forest Service's Northeastern Forest Experiment Station

In the mid 1970s, the US Forest Service developed an active urban forestry program in the northeastern US. Components of the program included hydrology, meteorology, economics, geography, recreation psychology, landscape architecture, and wildlife conservation. At the time, the Northeast was rapidly urbanizing and it was felt that people were losing contact with nature, natural processes, and wildlife in particular. Jack Ward Thomas, a research wildlife biologist, was then stationed at the Northeastern Forest Experiment Station at the University of Massachusetts and he focused much of his work on urban wildlife. He completed his PhD at the University of Massachusetts in 1973 with a dissertation titled "Habitat requirements for suburban songbirds—a pilot study" (Thomas 1973). Thomas was soon transferred to Oregon and Richard DeGraaf, also a research wildlife biologist, carried on and expanded the urban wildlife-habitat work (Fig. 2.8). Thomas and DeGraaf studied the relationship of foliage height diversity to bird species diversity in urban areas, a concept first described by MacArthur and MacArthur (1961). DeGraaf's research

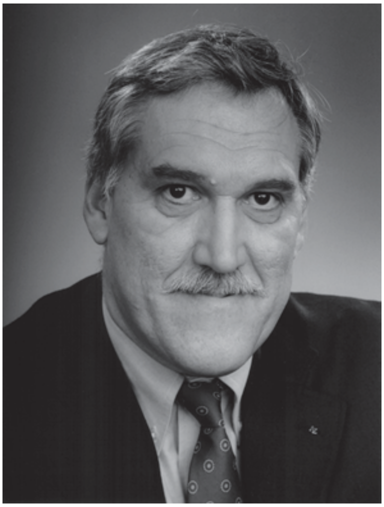

Fig. 2.8 As a US Forest Service research biologist, Dr. Richard DeGraaf focused his attention on urban bird-habitat associations from the mid 1970s through the early 1980s. His work helped to set the foundation for how landscape architects could incorporate good bird habitat in urbanizing areas

program continued to focus primarily on bird-habitat associations and considerable research was published through the early 1980s. A good overview of bird-habitat associations relative to landscape design was published in the proceedings of a national symposium on urban wildlife (DeGraaf 1987). That paper provided considerable information on how landscape architects could incorporate good bird habitat in urbanizing areas. In recognition of his work, Dr. DeGraaf was awarded the Daniel L. Leedy Urban Wildlife Conservation Award of 1991 by the National Institute for Urban Wildlife.

2.2.7 New York Department of Environmental Conservation

New York State's Department of Environmental Conservation started an urban wildlife program in 1976 (Matthews 1985). The first product of that effort was a survey of residents regarding urban wildlife (Brown and Dawson 1978). The study showed a high level of interest in, and a positive attitude toward, wildlife by New York's urban and suburban populations. Following the resident survey, a statewide inventory of seven urbanized areas was conducted to determine the availability of potential urban wildlife habitat. Data were used by municipal planners, developers, environmental organizations, and state agencies involved in the planning and development of metropolitan New York State. The Department also initiated an urban wildlife park program to provide opportunities for residents to enjoy an environmental educational experience in natural surroundings. Other efforts included production of educational materials on backyard wildlife, distribution of "shrub packets" to homeowners, and assistance in urban forestry through the State's Division of Lands and Forests.

2.2.8 *Other Programs and Activities in the USA*

2.2.8.1 Federal Government

At the federal level, the US Fish and Wildlife Service manages the National Wildlife Refuge System. Several of the refuges are located in urban areas, including John Heinz at Tinicum on the outskirts of Philadelphia, Pennsylvania; Bayou Sauvage, within the city limits of New Orleans, Louisiana; Minnesota Valley, within the metropolitan area of Minneapolis-St. Paul, Minnesota; and Don Edwards San Francisco Bay Refuge in California (US Fish and Wildlife Service 2012a). In 1999, the Service initiated its Urban Conservation Treaty for Migratory Birds to help municipal governments conserve migratory birds. The first two treaties were signed with New Orleans, Louisiana, and Chicago, Illinois. Seventeen other cities have been added as of May 2012 (US Fish and Wildlife Service 2012b).

In 1985, the National Park Service (NPS) renamed its Washington, DC-based Ecological Services Laboratory for the national capital region. The new name was Center for Urban Ecology. The Center was created to better describe the types of services that the region's natural science program provides. Among its activities are programs dealing with air and water resources, soils and agronomy, pest management, vegetation, and wildlife in NPS urban parks (National Park Service 2012).

2.2.8.2 State Government

Other states followed New York with designated urban wildlife programs. The Missouri Urban Biology Program was created in 1978 with three primary objectives: (1) to network information regarding natural history topics to the general public, (2) to assist public and private landowners regarding habitat management, and (3) to acquire significant habitats in or around metropolitan areas (Werner and Tylka 1984). The program also constructed urban nature centers in St. Louis, Kansas City, Springfield, and Jefferson City to provide outdoor-oriented educational opportunities to urbanites (Thorne and Witter 2001) (Fig. 2.9). Other early state programs included Washington in 1981 (Penland 1987) and Arizona in 1986 (Shaw and Supplee 1987). Additional states followed these early examples and the importance of state wildlife agencies focusing more attention in metropolitan environments is better recognized today.

2.2.8.3 Local Government

Although there is not a thorough study summarizing urban wildlife programs of local governments, the following examples illustrate what a few cities and counties are doing. In New York City, a new division, the Natural Resources Group, was created in the Department of Parks and Recreation in 1984 (Nilon et al. 1987). The

Fig. 2.9 The Missouri Urban Biology Program, created in 1978 by the Missouri Department of Conservation, was designed to better address needs of people in urban areas. Early efforts of the program included construction of the Powder Valley Conservation Nature Center near St. Louis (shown here) and Burr Oak Woods Nature Center near Kansas City. These facilities are still operational and urban nature centers have been added in other metropolitan centers in the state

group's primary focus is developing management plans for natural areas within city parks. Wildlife-related recreation and nature interpretation are important components of the plans. The Natural Resources Group also is involved with habitat restoration. New York City manages a 10,800-ha park system, 3600 ha of which are designated natural areas consisting of habitat fragments that are used heavily by people. Two types of restoration practices have been implemented: (1) restoration of degraded ecosystems and habitats, and (2) creation of new habitat. The Natural Resources Group restoration efforts focus on forests, meadows, freshwater wetlands, and salt marshes (Matsil and Feller 1996).

Portland (OR) Audubon Society initiated an effort to establish a metropolitan wildlife refuge system in the mid 1980s (Houck 1991). As the program evolved, it became evident that public interest was broader than just wildlife. In 1989, the regional planning body, known as Metropolitan Service District, which included the city of Portland and the surrounding tri-county urbanized area, instituted a regional natural areas program that evolved into the Metropolitan Greenspaces Program. The program published a report on regulatory and nonregulatory strategies for protecting open spaces for people and wildlife throughout the USA, with examples from South Africa, Canada, France, and New Zealand (Brooks and Wortman 1999).

Montgomery County, Maryland, which borders Washington, DC, established a natural resources planning and management program in the mid 1980s (Hench et al. 1987). The program elevated ecological concepts and principles within the local government. It has evolved over the years and now is called the Natural Resources Stewardship Section of Montgomery County Parks. Biologists there are involved with vegetation management, deer management, wildlife conflict resolution, and related matters (Montgomery Parks 2012).

Some other local jurisdiction efforts are noteworthy. In Ohio, the city of Toledo, through Toledo Metroparks, is restoring the historic structure to the 1495-ha Oak Openings Preserve Metropark in northwest Ohio (Abella et al. 2001). The city of

Boulder, Colorado, which owns and manages about 14,600 ha of open space and mountain parklands, has developed a forest management plan designed to restore variability in forest structure and disturbance processes that mimics historical processes to the extent possible and practical (Brown et al. 2001). Other examples of local jurisdictions successfully managing wildlife and natural resources include Howard County, Maryland (Howard County, Maryland 2012), Cook County, Illinois (Anchor 2009), Pitkin County, Colorado (Tennenbaum 2009), and Fort Worth, Texas (Denkhaus 2009).

2.2.8.4 Private Organizations

Other private organizations have focused attention on wildlife in metropolitan areas. The Trust for Public Land, a national land conservation organization founded in 1972, shifted greater attention to urban areas in 1994 with its Green Cities Initiative. That initiative was developed to better meet the park and open space needs of people residing in urban areas (The Trust for Public Land 1994). The Humane Society of the United States developed an urban wildlife program in the mid 1990s and operates an Urban Wildlife Sanctuary Program (The Humane Society of the United States 2012). The Fund for Animals and several other organizations have programs dealing with urban wildlife, particularly with regard to conflict resolution (Hadidian and Smith 2001).

2.2.9 International Programs and Activities

2.2.9.1 Europe

In 1980, the Second European Ecological Symposium, held in West Berlin, focused on urban ecology (Borkamm et al. 1982). That meeting attracted some 400 attendees from 18 countries. It brought together the limited information then available, encouraged further interest and research in urban ecology, and encouraged use of ecological inputs to the urban planning process.

The UK has been active in the urban arena for many years. Shenstone (1912) described the flora of building sites in London and later Fitter (1945) presented a natural history of that city (a similar work for New York City was reported by Kieran 1959). George Barker was the first urban coordinator of the Nature Conservancy Council, now Natural England (the UK government's nature conservation agency), and summarized European urban wildlife programs for a national symposium held in 1986 (Barker 1987). Barker also was instrumental in establishing the United Kingdom-Man and the Biosphere Urban Forum in 1990 (UK-MAB Urban Forum 2012). Dr. David Goode was particularly active in the London area (Goode 1991), first with the Greater London Council, and later with The London Ecology Unit, which he directed. He was instrumental in initiating a series of handbooks to draw attention to urban ecology issues. First in the series was *Ecology and Nature*

Conservation in London (Greater London Council 1984) followed by a guide describing how to create habitats in urban areas (Baines and Smart 1991). Additional handbooks were prepared on the wildlife of London boroughs. Peter Shirley was active in urban wildlife with the Urban Wildlife Trust in Birmingham and The Wildlife Trusts, a network of 47 local Wildlife Trusts working to protect wildlife in urban and exurban habitats throughout the UK (The Wildlife Trusts 2012). Dr. Steve Harris of the University of Bristol developed an early research program focused on urban red fox (*Vulpes vulpes*) ecology (Harris 1977, 1981; Harris and Rayner 1986).

Researchers in the Netherlands introduced the concept of "ecological landscapes" as a new approach to the design of urban open space in the late 1960s (Ruff 1987). Ecological processes such as plant succession and concepts such as the linking of diversity and stability were adopted as bases of ecological landscape design. Emphasis also was placed on use of native species, resulting in a less formal and more natural appearance. Following the lead of the Netherlands, landscape architects and urban conservationists in other countries became interested in the concept of "ecological landscapes." Notable in this regard is work in the UK (Baines 1985, 1986; Brookes 1998) and the USA (Diekelmann and Schuster 1982; Thompson and Steiner 1997; Link 1999).

Other European researchers and organizations have been working for some time to understand and enhance urban wildlife and natural resources. For example, in Germany, Dr. Herbert Sukopp and colleagues at the Technical University of Berlin devoted considerable effort to urban ecology, particularly plant communities (Sukopp 1990). In Poland, Dr. Maciej Luniak and colleagues of the Polish Academy of Sciences maintained a research program for many years, focused particularly on birds and invertebrates (Luniak 1990). Furthermore, The United Nations Educational, Scientific, and Cultural Organization's (UNESCO) Man and the Biosphere (MAB) Program, based in Paris, includes an urban ecosystems component (UNESCO 2012).

2.2.9.2 Other International Programs

Durban, South Africa created a Metropolitan Open Space System (D'MOSS) in the mid 1980s in an effort to improve the long-term quality of life for residents and retain some of the original plant and animal communities (Roberts 1994; Fig. 2.10). The plan was founded on the principles of island biogeography theory and consisted of core reserves, connecting corridors, and buffers. Core reserves conserving native plant and wildlife communities were linked by connecting corridors. Buffer areas were comprised of other open spaces, such as sports fields, golf courses, parks, cemeteries, private gardens, and road and rail rights-of-way. The Durban example is part of a growing trend worldwide in urban open space development and is focused on providing wildlife habitat as well as human recreation in such areas. Another example is the park connector network of Singapore that used principles of conservation biology and landscape planning to guide the implementation of a connected network of parks (Briffett et al. 1997, 1999, 2000).

Fig. 2.10 Durban, South Africa created a Metropolitan Open Space System (D'MOSS) in the mid 1980s. The plan was founded on the principles of island biogeography theory and consisted of core reserves, connecting corridors, and buffer areas. The effort was designed to improve the long-term quality of life for people and to retain some of the original plant and animal communities with continued development

In the 1990s, a focus on urban ecology emerged in Australia. Urban Ecology Australia, a non-profit, community-based urban environmental organization, was founded in 1991 (Urban Ecology Australia 2011) and in 1998, the Australian Research Centre for Urban Ecology was established (Australian Research Centre for Urban Ecology 2002). Furthermore, there is an active urban wildlife research program, the Suburban Wildlife Research Group, housed in the Australian School of Environmental Studies of Griffith University, Nathan (Rollinson and Jones 2002; Jones and Nealson 2003; Rollinson et al. 2003).

Conclusion: Reflection and the Future

In the mid 1970s when I got involved in urban wildlife just out of graduate school, Al Geis, Larry VanDruff, and Dick DeGraaf were in the early stages of their urban wildlife programs and were at the center of the urban wildlife field in the USA. All three were involved with The Wildlife Society's Urban Affairs and Regional Planning Committee and participated in the annual open exchange meetings held at the North American Wildlife and Natural Resources Conference. The field was small and we all knew one another. We were focusing on a neglected, but important, area of the wildlife field, where most people lived and interacted with wildlife on a daily basis. It was an exciting time to be a young wildlife professional working in a relatively new field. There was a lot of enthusiasm and promise for the future. The field, however, did not grow rapidly. As with all programs and activities, funding was a major driving force. Wildlife budgets were not flush with extra funds to devote to an emerging field. In the late 1970s, the Urban Affairs and Regional Planning Committee attempted to address this issue by drafting model federal legislation focused specifically on urban wildlife. Unfortunately, that legislation never advanced. At the time, work also was progressing on a "nongame" bill that was soon passed as the Fish and Wildlife Conservation Act of 1980. Over the years there were fluctuations in US Federal and State urban wildlife programs, and a gradual increase in local

Fig. 2.11 In modern development practice, considerable open space is set aside as urbanization expands. Urban wildlife biologists should be involved with making decisions on how such areas are managed. In recent years, more and more local jurisdictions have hired wildlife biologists to assist with this effort

programs. I believe the trend of greater local government involvement in wildlife conservation and management will continue.

Interestingly, two private industries have developed over the years focused largely in metropolitan areas. Wild Bird Centers of America, Inc.®, and similar businesses, cater to the public's interest in backyard bird feeding, and an industry focused on wildlife damage helps residents deal with animals causing nuisance or damage around the home. Two examples of the latter are Critter Control® and Humane Wildlife Services of The Humane Society of the United States.

With regard to the future, I believe we should consider wildlife in the urban environment as an important part of a broader picture, not as an end unto itself. Urban ecology, urban nature conservation, and urban biodiversity all have an urban wildlife component. In urban areas, wildlife should be a part of park management, open space management, street tree management, stormwater and watershed management, and even backyard management. Landscapes that are visually attractive to people can be created to benefit wildlife as well. Many landscape architects recognize audio aesthetics as well as visual aesthetics and bird song ranks high as a sound people enjoy.

Also looking ahead, I believe we need a better understanding of, and programs addressing, wildlife damage management (Chap. 17) and bird feeding, along with better knowledge of the impact of urbanization on wildlife (Chap. 7). Considerable urban green space will continue to be set aside and more thought should be given to restoring and managing this space (Fig. 2.11).

Fig. 2.12 Humans evolved in the natural world and perhaps there is an innate social need for wildlife and nature. Is this need expressed by the pets humans keep and such activities as wildlife watching and bird feeding that are practiced by millions of people? Medical research indicates that companion animals provide health benefits to people. Perhaps wild animals, including birds visiting bird feeders, also provide human health benefits when people enjoy them as wild animals

I think it would be helpful if we could better define a human "social need" for wildlife and nature. There is some evidence of this to date (Chap. 6). Research indicates that companion animals provide health benefits to humans (Friedmann et al. 1980; Barker and Wolen 2008; Friedmann and Son 2009; O'Haire 2010). Perhaps wild animals, including birds visiting bird feeders, provide human health benefits as well (Fig. 2.12). A "nature restoration hypothesis" has been proposed that incorporates the notion that natural views of trees and other plants tend to reduce human stress and anxiety (Ulrich 1979; Gold 1986). More recently, Grahn and Stigsdotter (2010) surveyed 733 adult Swedish residents of nine cities and towns in Sweden regarding human stress and urban green space. The authors concluded that urban green space should be viewed as a resource of importance to public mental health. Green environments may help to improve behavior of children with attention-deficit hyperactivity disorder (Kuo and Faber Taylor 2004). These studies indicate that wildlife and nature are not luxury value items that one can address only after meeting more pressing basic human needs. They suggest that wildlife and nature are, in fact, important basic human needs. More data of this nature will strengthen conservationists' appeals to governmental authorities to weigh wildlife and nature more heavily in decision-making processes.

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Chapter 3
Urban Wildlife Science in Coupled Human–Natural Systems

Michael W. Strohbach, Paige S. Warren and M. Nils Peterson

Keywords Coupled human and natural systems · Urban wildlife science · Human–wildlife interaction · Conceptual model · Ecosystem services · Feedback · Threshold · Resilience · Shrinking cities · Coyotes · Vultures

3.1 Introduction

Coupled human and natural systems (CHANS) are defined as systems in which human and natural components interact (Liu et al. 2007a). In most parts of the world, humans have altered whole landscapes for their needs, and in return, humans have adapted their culture to the specific challenges and opportunities of their environments. The cultural landscapes of Europe and other parts of the world are vivid examples of coupled human–natural systems (Naveh 1998). Such landscapes are the result of "the combined works of nature and of man" over centuries and reflect traditional land-use practices, are often biodiversity rich, and create a strong sense of place and recreational value for humans (World Heritage Convention 2012). Even regions thought to be pristine before the arrival of the Europeans, like North American forests or the Amazon River basin, are increasingly understood by scientists to be the product of human–nature interaction (Mann 2006). However, not until the twentieth century did human influence increase to such an extent that few ecosystems on earth remained pristine (Vitousek et al. 1997). For example, pollution from industrial areas in North America, Europe, and Asia caused and still causes acidification of soils and lakes in downwind regions and countries far away from

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M. W. Strohbach (✉) · P. S. Warren
Department of Environmental Conservation, University of Massachusetts—Amherst,
Amherst, USA
e-mail: michael.strohbach@geo.hu-berlin.de; michael.strohbach@gmail.com

M. W. Strohbach
Humboldt University, Department of Geography, Unter den Linden 6, 10099, Berlin, Germany

M. N. Peterson
Fisheries, Wildlife, and Conservation Biology Program, North Carolina State University,
Raleigh, NC 27695-7646, USA

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the source (Galloway 2001). Anthropogenic climate change is likely to impact the whole planet (Parry et al. 2007). In many environmental challenges, multiple ecological and social issues are interwoven to an extent that they cannot be addressed by traditional disciplinary science. Rather, they require inter- and trans-disciplinary approaches to research, conservation, and management (Palmer 2004).

Nowhere is the interaction of human and natural systems as intense as in cities (Kareiva et al. 2007), and therefore, the urban wildlife is distinguished from other wildlife by the increased level of interaction with people and human modified environments (Chap. 1). Today most humans live in urban areas and the world is on a trajectory for 67 % urban population in 2050 (United Nations Population Division 2012). Landscape change and invasive species associated with various forms of urban development are already the leading causes of species endangerment in the US (Czech et al. 2000). Wildlife science largely ignored CHANS in the past, even in relatively urban areas, because urban areas did not dominate the landscape and human–nature interactions were predominantly local (Liu et al. 2007b). Classical ecology explains wildlife-habitat relationships, behaviors, and demographic characteristics and how they are affected by abiotic factors (e.g., climate). As urban areas expand into previously rural environments, human behavior and decisions become an additional factor (Chaps. 7–10). For instance, urban raccoon (*Procyon lotor*) ecology is influenced by food supplies created by human behaviors, cover provided by green spaces, and interactions including human responses to property damage or rabies outbreaks associated with raccoons and raccoon responses to garbage disposal and land clearing. Small changes in one system can fundamentally change the interactions among systems. For instance, declining pet vaccination rates can intensify the feedback from raccoon rabies by placing humans at greater risk for disease exposure (Palamar et al. 2013). Efforts to understand urban wildlife ecology require explicit consideration of interactions and feedback between social and natural systems. Such consideration requires the measurement of ecological variables (e.g., diversity, landscape pattern), social variables (e.g., economics, social networks, values), and variables that link social and natural systems (e.g., anthropogenic landscape change, natural resource use, and waste disposal; Liu et al. 2007a).

The remainder of this chapter describes the dominant models developed for conceptualizing CHANS, and key principles for urban wildlife science that emerge from a perspective rooted in CHANS.

3.2 CHANS Models

3.2.1 Overview of Dominant CHANS Models

The idea that humans fundamentally influence natural systems on large scales did not arise until the nineteenth century, but became well established in the twentieth century (Goudie 2000). Although the phrase “coupled human and natural systems” did not appear in the scientific literature until the beginning of the twenty-first century, increasing attempts were made to promote interdisciplinary approaches to environmental conservation and management. One example is the *Man and the Biosphere Programme* (MAB) of the UNESCO established in 1971.

In the late 1990s, several influential papers on CHANS were published. Until then most studies did not recognize the links between the social, biotic, and abiotic environment and viewed them as independent systems, thus failing to recognize the complex interactions and feedbacks of parts of the systems (Pickett et al. 1997). Machlis et al. (1997) proposed the "human ecosystem" as an organizing concept for ecosystem management. In such a system, the human social system depends on a set of critical resources, including natural (e.g., energy, fauna, wood, or water), socioeconomic (e.g., labor or capital), and cultural resources (e.g., myths and beliefs; Machlis et al. 1997). Building on this work, Pickett et al. (1997) adapted CHANS for urban ecosystems, calling their integrated approach the "human ecosystem model." Naveh (1998) used the term "cultural landscape" to refer to CHANS. He proposed including the notion of "culture" into landscape restoration to broaden the conceptual and methodological scope to deal with complex human-ecological issues.

Research on CHANS was strongly facilitated by several national and supranational organizations like the US National Science Foundation's *Dynamics of Coupled Natural and Human Systems (CNH) Program* that started funding research in 2000 (The National Science Foundation 2012), the *pressure-state-response* (PSR) model by the Organization for Economic Co-operation and Development and its extension, and the *driving forces, pressures, states, impacts, responses* (DPSIR) model by the European Environment Agency (EEA 2012). Likewise, the *Long Term Ecological Research Network* (LTER) developed the *Integrative Science for Society and Environment* (ISSE) CHANS framework for guiding research (Taylor 2007).

All CHANS models have in common that they focus research on: (1) the patterns and processes that link human and natural systems; (2) reciprocal interactions and feedbacks between humans and the environment; and (3) understanding within-scale and cross-scale interactions between human and natural components (Liu et al. 2007b). Figure 3.1 shows a conceptual model of a CHANS with all its components and linkages that will be described and illustrated with urban wildlife related examples below.

3.2.2 The Components of CHANS Models

3.2.2.1 Ecosystem Services and Disservices

Ecosystem services are all the goods and benefits that humans receive from ecosystems and that constitute human well-being (Daily 1997). From the wood we use for paper making to the pollination of food crops, many aspects of our daily life are a result of ecosystem processes. In CHANS, ecosystem services represent a direct link between nature and society (Pickett et al. 1997; Fig. 3.1). To make that link

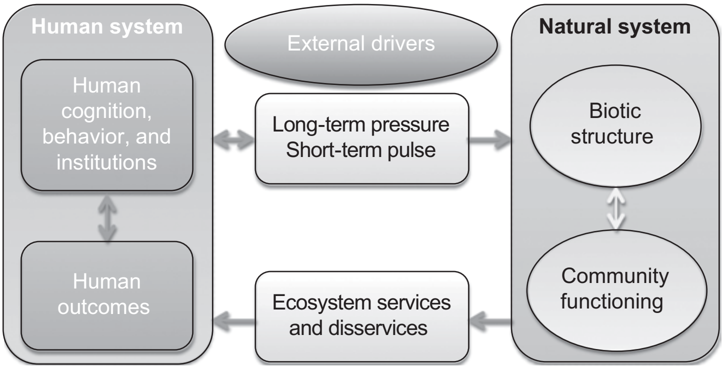

Fig. 3.1 A coupled human and natural system framework. (Source: Taylor 2007, modified)

comprehensible for human society, the services have to be quantified and valuated (Daily 1997). However, the valuation of ecosystem services is not easy. Values attributed to certain ecosystem services vary significantly between different cultures, social groups, and political parties, and valuation requires comparing and weighing different services against each other (e.g., water provision versus crop production). Economic valuation has made tremendous progress, as can be seen in *The Economics of Ecosystems and Biodiversity* (TEEB) report (Sukhdev et al. 2010). However, economic value only captures part of the total value of ecosystem services (de Groot et al. 2010). There are also sociocultural values (e.g., religious beliefs) that humans attribute to ecosystem services and that cannot necessarily be expressed in an economic value but must still be considered for conservation and management of ecosystems (see Box 1). A third evaluation domain has been developed by natural scientists based on the contribution of parts of ecosystems to the overall ecosystem health, for example the contribution of a particular tree species to erosion control or the facilitation of other species[1] (Millennium Ecosystem Assessment 2003).

Ecosystem services can be subdivided by the way they contribute to human well-being and there are various classification schemes. The *Millennium Ecosystem Assessment*, a highly influential report by the United Nations that informs about the state of earth's ecosystems and the consequences of ecosystem change on human well-being, subdivided ecosystem services into *provisioning* (e.g., food production from wildlife and fisheries), *regulating* (e.g., flood control by riparian forests), *cultural* (e.g., recreational value of wildlife) and *supporting* services (e.g., nutrient cycling). To stress the importance of biodiversity for the provision of ecosystem services, research was expanded by the concepts of *service-providing units* (SPUs) and *ecosystem service providers* (ESPs). SPUs are "the collection of individuals from a given species and their trait attributes necessary to deliver an ecosystem service at the desired level" (Harrington et al. 2010), for example the number of wild bees necessary to pollinate a particular crop (Luck et al. 2003). A certain popu-

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[1] This is closely linked to the ecosystem engineer concept introduced by Jones et al. (1994).

Box 1: Loss of ecosystem services from disappearing vultures

Ecosystem services often are not valued until the service provided is lost, as was the case with vultures in India. The uncharismatic and cross-culturally unlovable vulture (Subramanian 2011) provides the essential ecosystem service of consuming the decaying flesh of wild animals, livestock, and even humans. Three vulture species of the genus *Gyps* are resident in India, Oriental white-backed vulture (*Gyps bengalensis*), long-billed vulture (*G. indicus*), and slender-billed vulture (*G. tenuirostris*). All three declined across India by over 98% between 1992 and 2007 (Cuthbert et al. 2011). A clear culprit of the declines turned out to be an anti-inflammatory drug, Diclofenac, long used in people and then recently applied in livestock (Green et al. 2004; Oaks et al. 2004; Shultz et al. 2004). Widespread use of the drug in livestock produced a pulse of drug residue in the corpses of cattle. The drug kills the vultures that consume these corpses within days (Oaks et al. 2004; Cuthbert et al. 2011). Once its catastrophic effects were understood, the drug was banned across the region in 2006 (Ogada et al. 2012). However, evidence is mixed as to the effectiveness of the ban and potential for recovery of the vulture species (Cuthbert et al. 2011; Ogada et al. 2012).

The loss of the vultures has led to a series of surprises. Throughout India, concerns are being expressed over a wide range of effects that are rippling through the country’s social and ecological systems (Markandya et al. 2008; Subramanian 2011). Other scavenging birds appear to be less effective at removing carrion, leading to an accumulation of livestock carcasses at carcass dumps (Markandya et al. 2008; Subramanian 2011). Feral dogs, which also feed on carcasses, have increased by 30% over the period of vulture decline, with feared consequences for rabies and other diseases (Markandya et al. 2008; Ogada et al. 2012). India already had the highest rate of rabies infections in the world (Markandya et al. 2008). Other economic consequences include effects on bone collectors who collect bones of cattle after they have been cleaned by scavengers to supply the fertilizer industry (Markandya et al. 2008). Religious practices have also been affected. Members of the Parsi community believe that burning or burying their dead pollutes nature. They have lain their dead on “Towers of Silence,” where the corpses are devoured by carrion-eating birds for centuries (Fig. 3.4). Without vultures to remove the flesh from corpses, the Parsis in Mumbai have been forced to begin using solar concentrators on their towers to facilitate decay (Markandya et al. 2008).

The loss of the ecosystem services provided by vultures provides a potential feedback mechanism, increasing support for efforts to restore the vulture populations. A strong international response to the loss of the vultures has galvanized conservationists and led to a captive breeding program (Subramanian 2011; Ogada et al. 2012), calling attention to declines in other vulture species in Asia and Africa, where declines are due to factors different from the use of Diclofenac (Ogada et al. 2012). Currently, however, the vultures remain functionally extinct in India (Ogada et al. 2012). It remains unclear whether the catastrophic loss of the vultures constitutes a permanent shift in the socio-ecosystem.

lation size is needed to provide a particular service in a particular location and this must be considered in conservation. Closely related, ESPs include a wider set of ecosystem components, being defined as "populations, communities, functional groups, or trait attributes thereof, as well as abiotic components such as habitat type, that contribute to ecosystem service provision" (Harrington et al. 2010). In the aforementioned pollination example, the ESPs could be natural forest patches that provide habitat for wild bees that pollinate crops (Kremen 2005). The distinction between SUPs and ESPs is not always clear and there is a continuum between the two (Luck et al. 2009). While some services can be attributed to a SPU, like pest control in an apple orchard by a particular bird population, other services can only be provided by larger ecological units, for example water regulation by the whole forest vegetation (Luck et al. 2009).

Most often, city dwellers depend on ecosystem services imported from far away areas and that require ecosystems far larger than the city region itself (Folke et al. 1997, Chap. 1). Many services, however, are produced by the local urban ecosystems (Niemelä et al. 2010). Some of these services can only be produced by local ecosystems that remain relatively intact and complex, including wetlands that help treat wastewater (Cairns and Palmer 1995). Other urban ecosystem services, such as air filtration and local climate mitigation, can be linked to simple green spaces and even street trees (Niemelä et al. 2010).

Wildlife is recognized to play an important role in urban ecosystems (Cohn 2005), but concrete ecosystem services that are provided by wildlife rarely have been studied in cities. Ecosystem services provided by wildlife are primarily regulating and cultural services. As an example for regulating services, some arthropods control pest insects and pollinate plants in vacant lots and community gardens (Gardiner et al. 2013). In Phoenix, AZ, initial studies showed that birds exert top down control on arthropod populations, including such pests as aphids (Faeth et al. 2005), but subsequent work has not found this enhanced top down control effect (Bang et al. 2012). The ecologic and economic value of seed dispersal by Eurasian Jays (*Garrulus glandarius*) has been estimated in Stockholm, Sweden by Hougner et al. (2006). As an example of a cultural service provided by wildlife, the recreational value of green spaces appears to increase with greater bird diversity (Fuller et al. 2007; Lerman and Warren 2011; but see Dallimer et al. 2012). However, much work remains to identify the causal links. Observing wildlife in cities, especially birds, is important to the well-being of people, as is indicated by the large amounts of money that urban dwellers invest on bird feeding (Jones and Reynolds 2008). By providing educational opportunities urban wildlife provide another important cultural ecosystem service (Barnett et al. 2006).

The ecosystem service concept has been criticized for representing only the ecosystem functions that influence human well-being, while ignoring how ecosystems pose a variety of nuisances and threats to humans, so-called ecosystem disservices (Dunn 2010; Zeide 1998). One must only consider the negative impacts of pests like *Rattus spp*. (Sullivan 2005) to realize that ecosystem disservices should not be ignored in CHANS, especially in the context of wildlife. Examples for ecosystem disservices caused by wildlife are discussed in Chap. 17 of this book.

3.2.2.2 The Social System

The social template in CHANS models is typically influenced by ecosystem services and disservices (Fig. 3.1). For example, the preference of people for open and green space and the services it provides is reflected in household location choices (Peterson et al. 2008) and housing prices (Bolitzer and Netusil 2000). Interaction with urban wildlife may be viewed as a positive and aesthetically pleasing experience among some residents in urban areas (Peterson et al. 2002; Leong 2009). Similarly, diverse and plentiful urban wildlife can positively influence educational opportunities and even human health (Chaps. 1, 6). Changing ecosystem services (e.g., protection from diseases, scenic views) impact human outcomes, including risk exposure, education, recreation, health, and attitudes. In the aforementioned example, the preference of people for housing locations in more natural areas with ample open space can increase conflict over wildlife management (Chap. 17) and exposure to Lyme disease or West Nile virus (Bradley and Altizer 2007).

The human outcomes driven by improved or degraded ecosystem services, in turn, influence human behavior (Fig. 3.1). Human behavior is complex in its own right without considering interactions with biological systems via ecosystem services. In CHANS, behavior can be defined as an individual's conscious or involuntary action or reaction to their experiences. Of course, many times people make behavior decisions somewhat instinctively rather than thinking explicitly about them (Fazio 1995).

In urban areas, human behavior guides everything from housing type and location to leisure activities and consumption patterns (Petersen et al. 2007), and agents of human behavior can be a range of entities such as individuals, households, businesses, or governments (Warren et al. 2010). For instance, concerns about increased risk of exposure to wildlife related disease, road accidents, and damages to property have led to community-based deer management programs implemented through a democratic process in Groton, Connecticut (Kilpatrick and Walter 1997). Alternatively, individual householders may choose to fence their property or hire wildlife damage experts to trap and remove wildlife from their property (Chap. 17). The same concern about risk could promote urban policies banning wildlife feeding or feeding pets outdoors.

3.2.2.3 Long-Term Presses and Short-Term Pulse Disturbances Driven by Human Behavior

Human behavior creates two kinds of forcing functions that influence natural systems: slower cycling processes, termed presses (e.g., urban expansion) and short-term pulses or disturbances (Fig. 3.1). For example, personal landscaping decisions among residential householders have huge implications for urban wildlife conservation because private residents make management decisions for major portions of the urban land area (Breuste 2004; Grimm et al. 2008) and even influence vegetation cover on public lands near their homes (Zhou et al. 2009). By 2005, turf grasses constituted more than 16,380,000 ha in the USA, an area three times larger than that dedicated to maize production (Milesi et al. 2005). Furthermore, that area was expanding annually, with 23 % of new urban land area (675,000 ha per year) dedicated to turf grass (Robbins and Sharp 2008). These turf grass dominated landscapes tend to be relatively sterile in terms of wildlife habitat because they lack vertical and horizontal structure and the native plant species required for food and cover. Recent research suggests that given the choice, residents prefer wildlife-friendly landscaping with native plants over turf grass, but erroneously assume their neighbors would oppose innovative native plant landscaping (Peterson et al. 2012). Although conversion of natural and agricultural lands to turf grass is easiest to conceptualize as a long-term press on natural systems, the phenomenon also creates short-term pulse disturbances. For instance, fertilizer used to manage turf grass can create huge nutrient pulses in urban aquatic systems after rainfall (Bijoor et al. 2008; Zhou et al. 2009). Similarly, most suburban residents notice pulses in air and noise pollution on sunny weekend days when the lawnmowers come out.

Migration by people represents another critical human behavior impacting urban wildlife conservation. In the USA, a growing number of people move into communities where home and business developments intersect with endangered species conservation. This is partly because people are moving out of regions with low species endemicity and endangerment rates in the Northeast and Midwest and into endangered species hotspots in the South and West (Gutmann et al. 1998; Rutledge et al. 2001). However, this press has helped promote policy changes like the explosive growth in Habitat Conservation Plans in the USA, which allow for partial urban development of endangered species habitat while engaging the public in urban endangered species conservation (Peterson et al. 2004). Similarly, recent trends of reverse migration from urban to rural communities and suburban sprawl magnify human impacts on wildlife and create the need for development regulations in areas where they were previously not needed (Peterson and Liu 2008). Urban wildlife conservation regulations are still relatively rare, but have been implemented in forward thinking communities throughout the world. Chapter 2 describes the work of several influential groups engaged in management of urban wildlife, including the National Wildlife Federation, the Humane Society, the Trust for Public Land and the Audubon Society (see also Chap. 14).

Market-based approaches are used less frequently for urban wildlife conservation than in rural contexts, but environmental economists have recognized that urban centers have the financial capital needed to support ecosystem service markets. New York City's decision to invest in protecting the watershed providing its water rather than invest in expensive water purification technology is the paradigm case of urban investment in rural ecosystem services, but wildlife examples are emerging. For instance, residents of Jamestown, a community in Rhode Island, paid nearby hay farmers to delay harvesting hay for several weeks in an effort to protect bobolink (*Dolichonyx oryzivorus*) nesting habitat (Anderson 2007). In this case, urbanite affinity for wildlife also helped preserve open space and habitat for a host of other species which rely on early successional vegetation.

Box 2: Coyotes (*Canis latrans*) in Urban Areas

Unlike wolves, coyotes (Fig. 3.2) have successfully adapted to human dominated environments over the past 150 years. They have expanded from their traditional range, the Great Plains, to most of North America—from Florida to Alaska, quickly adapting to new habitat types and new food sources (Levy 2012). Increasingly, coyotes are extending their success to metropolitan areas, becoming the most controversial carnivore species in North American cities (Gehrt and Riley 2010). Urban coyotes show great adaptability, shifting their activity to the nighttime to avoid direct contact with humans (Grubbs and Krausman 2009), exploring a wide range of food sources (Gehrt and Riley 2010), and using urban infrastructure such as old pipes to raise their pups (Levy 2012).

From a CHANS perspective, the range expansion of coyotes constitutes a press on the natural system (Fig. 3.1). Coyotes that establish themselves in urban areas change the biotic structure and the community functioning of urban ecosystems. By preying on Canada goose nests and rodents, coyotes can provide a valuable ecosystem service to urban residents. However, there are conflicts, too, as human pets are vulnerable to attacks by coyotes, and especially house cats frequently are killed and eaten (Gehrt and Riley 2010). These conflicts might change human behavior, encouraging people to keep cats inside (Crooks and Soule 1999). Given the negative impacts that cats have on birds (Baker et al. 2010), more cats indoors might in return reduce impacts of another human-facilitated press on urban wildlife: exposure to domestic cat predation.

3.2.2.4 The Natural System

Ecosystems are complex and their properties cannot be predicted from their components alone (Odum and Barrett 2005). Urban ecosystems are characterized, at least in principle, by the same processes as natural ecosystems: energy flow (from primary producers to consumers and finally to decomposers); biogeochemical cycles of nutrients and matter; limiting and regulatory factors for individual species and species populations; interaction between organisms, shaped by competition, predation, mutualism and commensalism (Odum and Barrett 2005). However, these processes are highly modified in cities by long-term presses and short-term pulses from the human system, and urban ecosystems are often not self-regulating (Rebele 1994). Due to the introduction of new species and the exclusion of native species, cities often contain novel wildlife and plant communities (Kowarik 2011). Species might be confronted with predators to which they are not adapted (Chaps. 7–9), and others might expand their range due to new habitat types or food sources (see Box 2). It will be a future challenge to manage such "novel ecosystems" for ecosystem service provisioning (Hobbs et al. 2006).

Fig. 3.2 A coyote in the Ballona Wetlands, Los Angeles, California. (Source: Courtney McCammon)

3.2.2.5 External Drivers

External drivers in CHANS are aspects that are not, or only marginally, influenced by the social and natural components of the system. They include weather and climate but also extreme events like tsunamis or hurricanes. Acidification by industrial emissions, pollution (see Box 1), climate change, and biological invasion are external drivers for natural systems that are influenced by human activities. There are also external drivers for the social system: broad scale demographic drivers (population growth in most parts of the world); economic growth or decline (see Box 3); sociopolitical drivers (e.g., political systems, role of women, levels of education); cultural and religious drivers (shaping for example values or consumption behavior); science and technology (e.g., industrial production of fertilizer).

3.3 Principles for Urban Wildlife Science Emerging from CHANS

The complexity and the reciprocal feedback loops (Fig. 3.1) inherent to coupled human-natural systems together generate a suite of phenomena that should be considered when managing such ecosystems. Indirect effects of a management strategy to enhance one ecosystem service may propagate through the system in ways that generate negative effects on other services or generate disservices (Haase et al. 2012). Still, researchers seek ways to characterize and predict CHANS. A suite of phenomena have been described (Liu et al. 2007a, b), but most of the examples of these phenomena are derived from nonurban systems. We attempt here to provide

Box 3: Shrinking Cities

Globally, cities are expected to continue to grow (United Nations Population Division 2012). However, many cities in the industrialized world have been losing population, and whole countries are entering a new demographic stage of population decline rather than growth (Oswalt and Rieniets 2006).

From a CHANS perspective, general economic and demographic trends are external drivers on the human system that may lead to shrinkage and decline in cities. In shrinking cities, upkeep and usage of public and private spaces are reduced or phased out, and infrastructure, housing, and commercial and institutional land are abandoned. This eases the press of urbanization, allowing for more habitat area and natural dynamics like forest succession (Fig. 3.3; Kowarik 2011). The ecosystems that arise from these processes can be great sites for environmental education and recreation because they are close to where most people live (Kowarik and Körner 2005). However, unmanaged green space in cities can create a sense of unease and insecurity (Lorance Rall and Haase 2011), conflicts over wildlife management (Gehrt and Riley 2010), and threats from vector borne diseases (Reisen et al. 2008) that have to be countered by management actions.

some examples more directly relevant to urban wildlife conservation. Some examples are speculative and point to areas where additional research is greatly needed.

A key concept in the study of CHANS is that of *resilience*, the capacity of a system to retain or recover essential elements or functions after disturbance (Gunderson and Holling 2002). Human activities produce disturbances in natural systems, and natural disturbances can likewise impact human societies in sometimes catastrophic ways (e.g., Hurricane Katrina). Thus, important research questions include understanding both: (a) the factors that promote the resilience of wildlife populations and communities to the urban presses and pulses (e.g., persistent urban growth); and (b) how human societies respond to the disturbances generated by wildlife in urban settings. For example, what makes human communities resilient to impacts from outbreaks of wildlife-borne diseases? How do humans respond to the high densities of many urban species, particularly those with damaging impacts, such as pest insects?

The concept of resilience is not often invoked in urban wildlife studies. Yet, there is already an emerging body of work that addresses factors supporting more resilient wildlife communities in response to the press of urban growth. For example, studies suggest that landscaping with native plants supports more native bird and insect communities in residential areas, thereby retaining some key elements or functions of the system prior to the disturbance of urbanization (Daniels and Kirkpatrick 2006; Burghardt et al. 2009; Lerman and Warren 2011; Lerman et al. 2012). A large gap in existing literature, however, is in characterizing the temporal dynamics of urban wildlife populations and communities in already urbanized areas (Ramalho and Hobbs 2012). What urban forms promote resilience in these communities to stressors like drought, climate change, or urban redevelopment?

Fig. 3.3 An approximately 15-year-old successional forest in front of the former Karl Krause factory in Leipzig, Germany. (Source: M. W. Strohbach)

Reciprocal feedback loops complicate the discussion of CHANS and the factors promoting their resilience. For example, human activities can accelerate the expansion of disease-bearing insects like the mosquitos that carry West Nile Virus (Harrigan et al. 2010). Simultaneously, some of these same human activities may support the densities of so-called super-spreader bird species like the house sparrow (*Passer domesticus*) that are responsible for most of the West Nile Virus transmission (Kilpatrick et al. 2007; Hamer et al. 2009), thereby forming hotspots for disease outbreak in urban areas (Hamer et al. 2011). The human concern over the outbreaks may lead to mosquito control programs involving spraying pesticides, which could cause further shifts in avian communities, for example a reduction in breeding success of insectivorous house martins (*Delichon urbicum*; Poulin et al. 2010). Feedback loops may be positive as well. Actions by a group of citizens to restore an urban wetland or revitalize a local park may lead to greater presence of charismatic species like birds, butterflies, and dragonflies (Strohbach et al. 2013; Primack et al. 2000). This can in turn lead to greater public participation in additional restoration efforts, supporting these populations still further (Kobori and Primack 2003). Box 2 outlines another reciprocal feedback loop.

The dynamics of CHANS are frequently *nonlinear*, and characterized by *sudden shifts* from one state to another when *thresholds* are reached. Again, these dynamics rarely have been invoked in studies of urban wildlife, but are nevertheless likely to be occurring. Nonlinearities already have been described in some cases in patterns of diversity and abundance along urban gradients (McKinney 2002). In addition, city size may constitute a continuous variable with thresholds at which species are either unable to persist in urban areas or, conversely, are under sufficient selection to adapt to urban environments and colonize them. The likelihood of a previously nonurban species arriving in urban areas is thought to be a product of both intrinsic factors (e.g., species density in surrounding areas) and environmental factors (e.g., what proportion of the species' range is urbanized; Evans et al. 2010). Studies of the history of colonization of urban areas by birds in Europe find nonlinearities and sudden shifts associated with the establishment of previously nonurban species in cities, but the causal factors are not always clear (Evans et al. 2010). Prior to 1820, the Eurasian blackbird (*Turdus merula*) was absent from cities, but once it colonized two cities in southern Germany, the species spread rapidly to others nearby (Evans et al. 2010). By contrast, for two other species, wood pigeon (*Columba palumbus*) and magpie (*Pica pica*), spatial proximity seems to have no relationship to the pattern of colonization (Evans et al. 2010). Likewise, in the USA, coyotes (*Canis latrans*) seem to avoid some urban areas (Champaign, IL, human population of metro area 231,891; 2010 US Census; Lavin et al. 2003) while thriving in others (Box 2; Chicago, IL, human population of metro area 9.46 million, 2010 US Census, and Southern California metropolitan region, largest metropolitan region in the USA; Gehrt et al. 2009; Ordeñana et al. 2010, respectively). One possibility is that a metropolitan area needs to exceed some threshold size before enough rural coyotes overcome their aversion to human disturbance to colonize the city.

Temporal dynamics are an important and not well-understood aspect of urban ecosystems. As with other CHANS, cities may exhibit *time lags* in phenomena such as extinction debts in response to habitat loss or degradation. For example, newer cities seem to retain greater native species diversity than older ones (Duncan et al. 2011; Aronson et al. 2014). One possible explanation is that an extinction debt has accrued in the newer cities, with species persisting but declining in population size. Alternative explanations include differences in urban form between older and newer cities and in the amount of green space they retain (Aronson et al. 2014). Other time lags include delays in the appearance of effects from changes in land policies (Liu et al. 2007a) or lags in the effects on wetland species from road construction (Findlay and Bourdages 2000).

History plays an important role in cities as in other coupled systems. The contemporary physical structure of a city is a product of layers of historic phases of design, policy, and human migrations (Warren et al. 2010). Conditions and processes in the past may have ongoing *legacy effects* on contemporary conditions (Liu et al. 2007a, b). Ecological legacy effects have mainly been described in many nonurban systems, but are increasingly described for urban environments as well. Current patterns of tree density and canopy cover in Baltimore, Maryland are thought to be legacies of stewardship activities of prior occupants (Troy et al. 2007). In residential soils in Phoenix, Arizona, the legacies of prior agricultural land use have had enduring effects on nutrient composition, detectable in some cases 40 years after urbanization has taken place (Lewis et al. 2006). Human waste deposition in cities may likewise have enduring effects on soils, detectable even centuries later (Davidson et al. 2006). These soil nutrient legacies are likely to have significant effects on soil arthropods that may then be propagated throughout the food web.

Heterogeneities in the coupling of human and natural systems occur at many scales (Liu et al. 2007b). Cities of different sizes and histories harbor different levels of biodiversity within their boundaries (Duncan et al. 2011; Aronson et al. 2014), and cities are expanding at different rates across the globe, with impacts on biodiversity projected to increase (Seto et al. 2012). Cities themselves are among the most heterogeneous environments on the planet (Pickett et al. 2011). Wildlife habitat and animal communities likewise vary along gradients of urbanization and with many other socioeconomic factors (McKinney 2002; Warren et al. 2010). Human engagement in wildlife oriented human activities, like bird feeding, is likewise heterogeneous within and among cities (Lepczyk et al. 2012). Even the nature of the coupling between social dynamics and urban environments are heterogeneous. For example, crime rates are generally lower in areas with greater tree cover in Baltimore, Maryland, but in some portions of the city, crime rates instead increase with higher tree cover, often in areas with greater land abandonment (Troy et al. 2007). This has implications for residents' willingness to support the preservation of trees and green spaces.

The complexity of CHANS can make it difficult or impossible for people to understand the linkages between the many interconnected processes. Managing for one outcome often leads to unpredicted and even perverse outcomes, or *surprises*. Policies intended to restrict urban growth, for example, can unintentionally lead to increased sprawling development outside of growth management boundaries (Liu et al. 2007a). When the ecosystem services provided by urban wildlife are not understood, the impacts of species loss also can be surprising (Box 1). Merely the persistence of some species in urban areas may be considered a surprise (Pickett et al. 2008), such as the population of red-backed salamanders (*Plethodon cinereus*) persisting in a green space "island" in Cleveland, Ohio (Walton et al. 2006). As urban ecological research becomes ever more sophisticated, addressing underlying processes instead of documenting patterns (Shochat et al. 2006), the complexity of urban ecosystems will undoubtedly lead to the revelation of additional surprises.

Conclusion

The complexity of human societies, of urban forms, and of the dynamic relationships between humans and their surrounding biota make cities particularly challenging to study. Any attempt to understand the spatial and temporal dynamics of urban wildlife populations and communities is incomplete without recognition that they are embedded in these complex coupled systems. Conceptual models of CHANS (e.g., Fig. 3.1) provide valuable heuristic tools for understanding the components of urban ecosystems and the interconnections within these ecosystems and with surrounding environments. The lessons of history tell us that managing urban ecosystems without recognition of their complexity is a perilous endeavor.

Fig. 3.4 Parsi "Tower of Silence" in Mumbai, India, around 1900. Vulture numbers have declined dramatically in recent decades, effecting also religious practices. (Source: http://www.gutenberg.org/files/27260/27260-h/27260-h.htm#Page_128, East of Suez: Ceylon, India, China and Japan published in February 1907)

The principles of CHANS provide a starting place for pursuing an understanding of how the coupled nature of urban systems influences wildlife. Our brief summary of these principles has revealed many avenues where additional research is needed to illuminate the dynamics of wildlife in urban systems.

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Chapter 4
Abiotic Drivers of Ecological Structure and Function in Urban Systems

Kirsten Schwarz, Dustin L. Herrmann and Melissa R. McHale

Keywords Abiotic drivers · Land cover · Land use · Water · Soil · Climate · Energy · Pollution

4.1 Introduction

The interactions between abiotic (nonliving) and biotic (living) factors define ecosystem ecology, and form an essential foundation for any effort to understand urban wildlife. The study of ecosystems considers the flow of energy and materials between organisms and their physical environment (Chapin et al. 2002). For instance, how much energy enters an ecosystem through the process of primary production—the conversion of CO_2, water, and solar energy into biomass—by plants significantly influences how many organisms an ecosystem can support. The interactions between abiotic and biotic components of urban systems are both fundamental to understanding the ecological functioning of cities and are used to explain wildlife-related phenomena, including trophic cascades, biodiversity, and species distribution across the landscape.

Cities, however, represent a unique ecosystem type that is highly influenced by human activities. Although cities contain traditional abiotic factors (e.g., water, soil), they also contain human-generated abiotic factors (e.g., buildings, roads) that are often highly engineered, managed, and dynamic (Fig. 4.1). Furthermore, abiotic characteristics that we might consider "natural" are modified or manipulated to an extent not experienced in nonurban systems. For example, humans move soil in

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K. Schwarz (✉)
Department of Biological Sciences, Northern Kentucky University,
Highland Heights, KY 41099, USA
e-mail: schwarzk1@nku.edu

D. L. Herrmann
Department of Plant Sciences, University of California, Davis,
Davis, CA 95616, USA

M. R. McHale
Department of Forestry and Environmental Resources,
North Carolina State University, Raleigh, NC, 27695, USA

R. A. McCleery et al. (eds.), *Urban Wildlife Conservation: Theory and Practice,*
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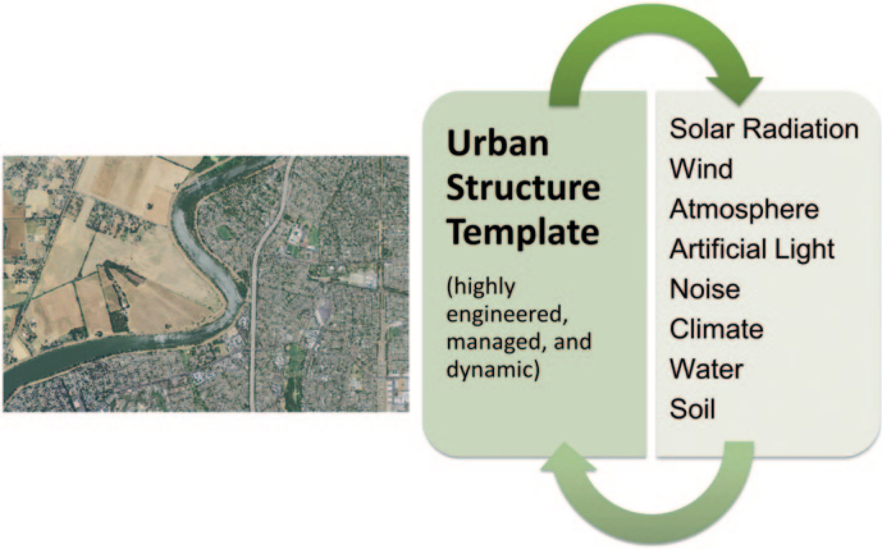

Fig. 4.1 The process of urbanization brings dramatic structural changes, demonstrated by the aerial photograph of Sacramento, CA showing the contrast between agricultural and urban land. Changes to the urban structure template drive changes to additional abiotic factors, including solar radiation, wind, atmosphere, artificial light, noise, climate, water, and soil. These, in turn, can have important feedbacks that change the urban structure template

large quantities during development, make soil susceptible to erosion and compaction, and introduce and remove nutrients. The manipulation of abiotic factors and addition of human-generated abiotic factors can have direct effects on the distribution of urban wildlife as well as their movement, diet, and reproduction (Ditchkoff et al. 2006).

Another unique aspect of cities is discovered when examining energy and material (e.g., water) flows associated with cities. Urban areas concentrate resources from large geographic extents and greatly increase the intensity of energy consumption, material inputs, and waste streams within a limited geographic extent (Decker et al. 2000). As a result, the spatial area influenced by human activities within urban areas often exceeds the spatial boundaries of those ecosystems (Chap. 1). For these reasons, understanding humans as drivers of ecosystem structure and function is important for all ecosystem types, but essential to our understanding of cities as ecosystems.

This chapter focuses on abiotic factors that can influence wildlife in the most obvious ways (e.g., pigeons roosting in crevices in buildings), as well as those processes we might not expect (e.g., changes in the tonal variation of mating songs among urban bird species). We first define urban in terms of land cover—what aspects of the urban environment make an area uniquely “urban” and how does it differ from its nonurban counterparts. Next we explore the flux of energy within cities and how the combination of solar radiation and urban pollutants creates a unique urban atmosphere and climate. This section also addresses wind, artificial lighting, and noise as abiotic drivers of wildlife ecology in cities. Water is explored from two perspectives: (1) the effect that urbanization has on the quantity, quality, and distribution of water, and (2) the effect that urbanization has on the way in which wildlife ecologists study water. Urban soil—the foundation of many abiotic-biotic interactions—maintains many of the same ecological functions as nonurban soils. Many of those functions—especially its role as a growth medium—are essential to urban wildlife. Finally, the unique nutrients and pollutants that are found in urban soils are addressed.

4.2 Urban Structure—Land Use and Cover

Urban areas often are described in terms of land use—literally how humans use the land. For instance, in an urban area you can expect to see residential, industrial, transportation, recreational, or vacant land uses. Sometimes the term “urban” itself is described as a land use, and is distinguished in contrast to agricultural, conservation, and even suburban areas. The dominant land uses in urban landscapes reflect the central functions cities serve: places for humans to live, work, and recreate. Land *use* only describes the basic activities that occur in one part of an urban area. In contrast, land cover describes landscapes in terms of the features that are present—trees, buildings, and impervious surface. Land cover does not describe how the land is used (i.e., its function), but instead it simply describes what is there (i.e., its structure). For example, a single family home may be described as residential using a land use classification. Using a land cover classification, the parcel would be described in terms of the amount of tree, building, grass, and impervious surface cover. A more robust analysis of urban landscapes is based on land cover which allows wildlife ecologists to test the relationships between ecological structure and function (Cadenasso et al. 2007).

Understanding urban land cover is fundamental to our understanding of cities as ecosystems, and therefore, much research has been dedicated to understanding what influences the distribution of these features across human-dominated landscapes (Grimm et al. 2008). At a broad scale, the planning departments of local and regional governing agencies direct the relative distribution of land use types in cities (Alberti et al. 2001). The actual structure of individual neighborhoods, though, may be determined by zoning codes or development companies and related to the social structure of the neighborhood (Grove et al. 2006; Cook et al. 2012). Furthermore, lifestyles and landscape preferences of people may have the most influence on cover at fine scales (e.g., Grove et al. 2004). As an important note, this nested set of drivers summarizes research on cities in regions or countries with advanced economies. Drivers may be very different in regions undergoing rapid rural-to-urban transitions or having less developed economies.

The most visually striking—even defining—features of the process of urbanization are associated with land cover change. Just as agriculture and forest lands appear to be relatively large and continuous extents of homogenous vegetation and bare earth, urban areas can also appear homogeneous from a distance. However, after closer examination, cities are actually a patchy mosaic of vegetation, bare earth, impervious or 'paved' surfaces, and buildings (Cadenasso et al. 2007). Transitions between land covers occur at the submeter level and multiple land covers can exist within a single land parcel. These characteristics of land cover are particularly important to ecologists that study the links between ecosystem structure and function, and require novel land cover mapping approaches. For example, if a wildlife ecologist was interested in relating bird abundance and distribution to urban tree canopy cover, new approaches to land cover classification that use an object-based approach may capture street trees in the landscape, which could be important to urban bird populations. Coarser scale pixel-based approaches would likely miss this potentially important habitat characteristic (Moskal et al. 2011).

As humans build cities, they fundamentally change the structure of the landscape by adding and modifying infrastructure. Urban infrastructure can be described as gray, green, or brown. Gray infrastructure encompasses the impervious surfaces of cities—mainly buildings and roads. Green infrastructure refers to vegetation in the city, including grasses and trees. Soil represents the brown infrastructure of cities. Changes to the gray, green, and brown infrastructure of cities are often depicted by urbanization gradients, where differences among urban and rural land uses are defined by the amount of imperviousness in the landscape—or changes to the gray infrastructure. Research using urbanization gradients clearly shows that changes to the structure of the system can translate to changes in its function (McDonnell et al. 1993). For example, the diversity of wildlife changes dramatically with the transition from rural to urban landscapes in response to changes in abiotic and biotic structure—the foundation of wildlife habitat (McKinney 2002). Wildlife can also respond to local or within-city variation in land cover and land use (Herrmann et al. 2012; Fig. 4.2).

4.3 Energy, Atmosphere, and Climate

Changes to land cover due to urbanization alter or redirect flows of naturally occurring energy, such as sun (i.e., solar radiation) and wind. Urban land uses also introduce materials and energy into the urban environment. Introduced energy and materials include heat, particulates, and reactive gases to the atmosphere (e.g., from car emissions), noise (e.g., traffic), and artificial lighting (e.g., street lamps). Urban land cover and land use also indirectly affect climatic conditions in and around cities. Overall, wildlife in cities encounter multiple environmental factors that differ from nonurban landscapes with the potential to affect the way wildlife move, live, and interact in cities.

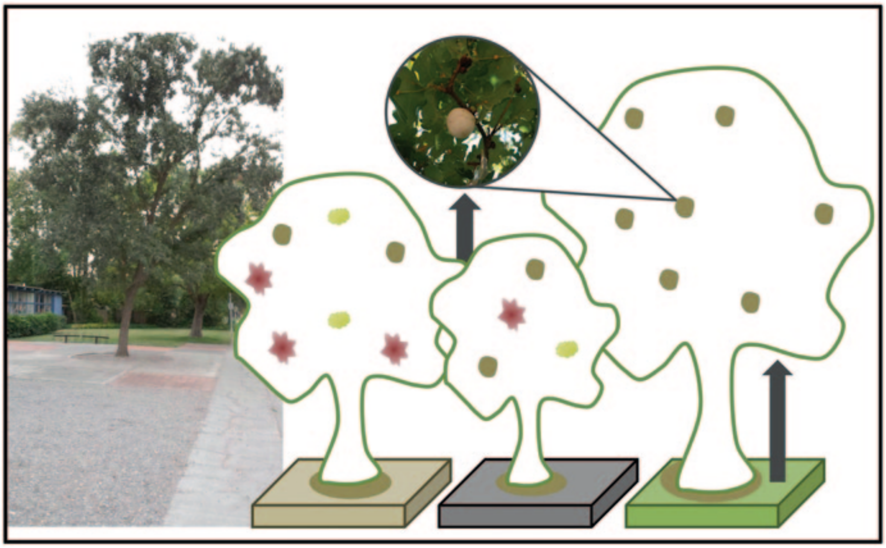

Fig. 4.2 Abiotic drivers in urban systems have important feedbacks to the biotic community and can affect the abundance of individual species as well as the species richness and evenness of wildlife communities. In this example, the land cover template (e.g., pavement, natural litter, or lawn) affects the diversity and abundance of the gall wasp community that the habitat—Valley oak (*Quercus lobata*)—supports

4.3.1 Solar Radiation

Additions of built structure and changes to vegetation—namely tree canopy cover—from urbanization can radically alter the distribution and character of sunlight. Buildings and tree canopy can block, reflect, and absorb sunlight, affecting the amount of solar energy reaching the ground, similar to a forest understory (Hutchison and Matt 1977). The distribution of solar radiation will affect plant growth, and the availability of energy in the form of heat. Therefore, urban structural changes, such as the addition of tall buildings, can affect resource availability, habitat structure, and access to necessary thermal conditions for wildlife. Bee and butterfly species diversity and abundance in New York City gardens, for example, was strongly determined by sunlight availability; buildings and tree canopy greatly reduced the amount of light available for maintaining floral resources and meeting thermal requirements (Matteson and Langellotto 2010).

4.3.2 Wind

Urban areas typically experience decreased wind speeds relative to rural areas (Kuttler 2008). However, depending on the geographic and environmental characteristics associated with the setting of the city, wind speeds may actually increase (e.g., Seoul, South Korea; Lee and Baik 2010). Within a city, wind speeds may be quite variable depending on urban structure. Infrastructure can lead to the creation of wind tunnels (e.g., canyons created by building lined streets), increasing wind speeds quite dramatically, or alternatively, the leeward sides of buildings may dramatically reduce wind speeds.

Wind can play important roles in shaping urban ecological systems. Wind is a factor in plant structure and growth (Ennos 1997), and decreased wind speeds due to urbanization have been shown to increase primary productivity (Bang et al. 2010). Wildlife can be directly affected by wind speeds; wind can affect thermal regulation, movement, and predator–prey interactions (Møller 2013). In one case, greater urban wind speed reduced insect abundance leading to lower reproduction and survival in an insectivorous bird community (Møller 2013). There can also be effects of wind on habitat creation for wildlife. For example, tree cavities are a critical habitat component for wildlife worldwide. These cavities are often created by fire in rural areas, but lack of fire and strong urban winds make wind-induced branch breakage a more critical driver in urban areas (Harper et al. 2005). Overall, wind may play a major role in structuring wildlife populations in urban remnants that are more exposed than nonurban habitat patches (i.e., greater influence of edge effects).

4.3.3 Urban Atmosphere

Urban activities modify atmospheric chemistry, and as a result there are increases in carbon, nitrogen, ozone, particulate matter, and metals in more urbanized regions (Gatz 1991; Idso et al. 2001; Kaye et al. 2006). Across cities, transportation is a major source of inputs to the urban atmosphere. Combustion of fuel for transportation and friction wear on vehicle parts (e.g., brake pads) combine to generate all classes of pollutants listed above. Industrial activity is another major source of urban air pollution; however, the type and intensity will vary across cities by specific activities, energy sources (e.g., coal or hydropower), and regulations on emissions. High atmospheric pollutant loads also lead to elevated deposition of pollutants in and near urban areas (Gatz 1991; Azimi et al. 2005), with consequences for terrestrial and aquatic wildlife. As will be discussed below, there are many ecological and evolutionary responses to urban air pollution (McIntyre 2000; Gregg et al. 2003).

4.3.3.1 Carbon

There are two major carbon (C) forms that affect most cities—carbon dioxide (CO_2) and organic C. First, “urban CO_2 domes” are formed from local fossil fuel combustion; levels of atmospheric CO_2 in cities can reach >600 ppm and regularly exceed 400 ppm, whereas background levels are <400 ppm (Idso et al. 2001). Elevated atmospheric CO_2 can have indirect consequences for wildlife through effects on plants. For instance, elevated levels of CO_2 can positively impact primary productivity via greater water and nutrient use efficiency with an accompanying change in leaf tissue chemistry (Leakey et al. 2009). Foliar chemistry generally shifts to a lower nutrient concentrations with negative consequences for herbivore populations and wildlife food webs mediated by herbivore dynamics (Stiling and Cornelissen 2007).

Organic carbon compounds—some of which are precursors to ozone formation—are elevated in the atmosphere due to industry, transportation, and other urban activities, such as cooking and heating (Azimi et al. 2005). Through the release of volatile organic compounds (VOCs) vegetation can also contribute to ozone formation (Fehsenfeld et al. 1992, see *Ozone* below).

4.3.3.2 Oxidized Nitrogen (N)

Transportation and other fuel burning urban activities result in the emission of oxidized forms of N (NO, NO_2, and NO_3, often collectively referred to as NOx) which are highly reactive in the atmosphere. Combined with VOCs in the presence of sunlight, NOx can form ground-level ozone (O_3, see *Ozone*). In addition, they are precursors to the formation of multiple secondary air pollutants (see *Particulate Matter*). Finally, oxidized N can be deposited on surfaces, in wet and dry forms; highly urbanized regions experience high levels of oxidized N deposition (Kaye et al. 2006). Elevated N deposition can decrease N retention by terrestrial ecosystems (Aber et al. 2003); increased N loss in urban uplands (i.e., terrestrial system) can lead to N pollution of receiving water bodies. Excess N reaching water bodies is associated with eutrophication conditions in lakes and coastal waters as well as NO_3 levels toxic to wildlife in freshwater lakes and streams (Fenn et al. 2003). The connection between urban uplands and aquatic systems is discussed in the *Water* section later in the chapter.

4.3.3.3 Ozone

Ozone is a main component of smog—the photochemical haze that forms in some urban regions. It is formed through a temperature-sensitive chemical reaction between VOCs and NOx in the presence of sunlight. Although the precursors, NOx especially, are largely formed in densely urbanized areas, ozone is often elevated in areas adjacent to urban centers due to scavenging activity—in which NOx and O_3 quickly react to form N_2O and O_2—under the high levels of atmospheric NO in the urban core (Gregg et al. 2003). Importantly, ozone concentrations exhibit distinct seasonal and diurnal patterns—elevated levels during the summer and afternoon hours (World Health Organization 2003). Ozone can play a significant role in reduced vegetation growth in urbanized regions (Gregg et al. 2003), and it is known to have negative effects on respiratory systems in humans and lab animals (World Health Organization 2003).

4.3.3.4 Particulate Matter

Urban suspended particulate matter, also called atmospheric aerosols, consists of directly emitted particles (primary aerosols) and chemically formed gas-particle reactions (secondary aerosols; Marley and Gaffney 2005). Human activities, particularly related to industry, transportation, and construction, can generate high levels of primary aerosols as well as the gases (e.g., NOx, SOx) involved in secondary aerosol creation. Coal burning is also a major source of aerosols in the cities near power plants lacking strong emission controls (Santosa 2010). Two size classes defined by the US Environmental Protection Agency are most often recognized: inhalable particulates (PM_{10}; <10 μm diameter) and respirable particulates ($PM_{2.5}$; <2.5 μm diameter). In the human respiratory system, PM_{10} get caught in the nose and throat, whereas $PM_{2.5}$ can reach the lungs (Santosa 2010). Size classifications, however, do not reflect the chemical nature of particles; additionally, ultrafine particles (nanoscale) can pose unique health hazards, but they are clumped with the $PM_{2.5}$ classification (Marley and Gaffney 2005). In particular, exposure to fine-particulate matter over several years is linked to lung cancer in humans (Pope et al. 2002).

4.3.3.5 Metals

Sources of metals in the urban environment include dusts from industrial products (e.g., paint) and friction surfaces (e.g., automobile brake linings) as well as combustion (e.g., leaded gasoline). Common contemporary and historical metals carried in the urban atmosphere at high concentrations include Pb, Zn, Cu, Al, Fe, and Ni (Pouyat and McDonnell 1991; Azimi et al. 2005; Schwarz et al. 2012) with many more that can potentially be present at elevated levels (Liu et al. 2003). Typically, heavy metals do not travel far from the point of emission before being deposited. Heavy metal accumulations were greatest in urban forest soils and decreased with distance from the urban center (Pouyat and McDonnell 1991). In addition, redistribution of heavy metals by the atmosphere results in spatially heterogeneous soil accumulation within urban areas based on proximity to source (Schwarz et al. 2012). Metals, along with other aerosols, can also be "scrubbed" from the urban atmosphere by building surfaces (Liu et al. 2003) and trees (Weathers et al. 2000; Tomašević et al. 2004). Wildlife is most likely to be affected by metals after they have accumulated in urban soils. Heavy metals can enter the food chain through bioaccumulation in terrestrial invertebrates or transported into aquatic systems (see *Soils*).

4.3.4 Artificial Light

Artificial lighting is an important abiotic driver of ecological interactions in urban systems (Longcore and Rich 2004). With the advances in electric lighting over the last century, artificial lighting has become widespread in urbanized areas, significantly increasing light levels in the night sky (Elvidge et al. 2001). Nighttime outdoor lighting is important to human well-being. It contributes to vibrant urban life as it helps create a sense of safety and security and facilitates human activities. However, it may have negative effects on wildlife populations. Of note, bird strikes on lighted tall buildings during nighttime migration is a significant source of mortality, especially for songbirds (Evans Ogden 1996).

The effects of nighttime lighting on ecological communities are more widespread than a direct driver of bird mortality. Artificial night lighting can affect wildlife physiology (e.g., melatonin production) and behavior, as well as community interactions (i.e., competition, predation; Rich and Longcore 2005). Bats, for example, exhibit different behaviors depending on the species (Stone et al. 2009, 2012). Fast-flying bat species benefit from increased insect abundance around artificial lighting. Slow-flying bat species, however, avoid lighted areas, presumably to avoid detection by predators. Importantly, ecosystem types within or near the urban matrix will be differentially vulnerable to the effects of nighttime lighting. Grasslands and deserts are especially affected because their open physical structure permits lighting to spread over a large area. Aquatic systems are also vulnerable as fish, zooplankton, and amphibian behavior and community interactions are strongly organized by light availability and light cycles (Moore et al. 2006; Perry et al. 2008; Becker et al. 2013). Alternatively, light color can be manipulated for ecological benefit. For example, red lights have limited penetration of water and therefore have less of an impact on aquatic ecosystems than standard lighting (Becker et al. 2013).

Presence or absence of artificial lighting is not the only consideration regarding its effects on wildlife. The distribution of lights within urban areas, their type and design, and lighting intensity or luminosity can be important to ecological interactions (Rich and Longcore 2005). For example, insect groups are known to be differentially attracted to conventional mercury-and sodium-based lighting and are affected by light outside the visible spectrum (Eisenbies and Hänel 2009). How lighting type will affect wildlife is especially important as new lighting technologies are adopted. Modern lighting (e.g., LEDs), importantly, uses a broader spectrum of lighting frequencies. This shift has the potential to dramatically reorganize ecological communities, particularly by affecting interspecific interactions. This can arise because wildlife taxa benefit differentially from the new lighting types. The ability of mammals and birds, in particular, to detect objects will improve much more than that of arachnids, insects, and reptiles (Davies et al. 2013). Behavior may also be affected due to improved vision. In one documented case, great tits (*Parus major*) increased the feeding rates of their young when exposed to LED lighting (Titulaer et al. 2012).

4.3.5 Noise

Urban areas contain multiple sources of anthropogenic noise. Noise associated with road traffic is the most significant source, but other notable sources include industrial activity, railroads, and outdoor condensing units (Warren et al. 2006; Barber et al. 2010). Patterns in the activity levels of urban noise sources create spatial and temporal heterogeneity in the urban soundscape (Warren et al. 2006). Noise decreases with distance from heavily trafficked roads leading to noise contrasts across the urban landscape. Built structures and vegetation can create fine spatial heterogeneity by blocking, reflecting, or absorbing sound waves. Diurnal patterns in noise exist as well. Of significance, noise levels increase around morning and evening commute times. As vehicular noise dominates the urban acoustic environment, anthropogenic noise will generally be composed of low frequency sounds. Finally, the noise pollution faced by wildlife in cities has analogs to conditions under which species evolved and encounter in habitats free of anthropogenic noise pollution. For example, noise in the city can be similar to a cacophony of competing calls in nonurban areas. Furthermore, the built environment can create acoustics mimicking those created by canyon structure acoustics (Warren et al. 2006).

Researchers are starting to understand and test the extent and pathways of how noise impacts wildlife (Kight and Swaddle 2011). Noise is a stressor that can have multiple behavioral, psychological, and physiological effects on wildlife (Barber et al. 2010), with strong evidence of its effects on humans (Passchier-Vermeer and Passchier 2000). Urban acoustics studies on wildlife generally have focused on the calling environment and concerns for how species are adapting and what the consequences are for fitness (Rabin and Greene 2002). Adaptations of wildlife to urban environments through behavioral responses to noise environment are frequently reported, especially in calling noises for birds (Slabbekoorn and Peet 2003), amphibians (Parris et al. 2009), and insects (Lampe et al. 2012). Wildlife have responded to noisy urban environments by changing the amplitude, frequency, and timing of calls (Warren et al. 2006). There is also a concern that noise could inhibit activities such as foraging, mating, and alertness to predators (Chan and Blumstein 2011).

4.3.6 Urban Climate

Common climate properties of cities emerge from an interaction of multiple changes associated with urbanization. An almost universal change is the creation of the Urban Heat Island (UHI)—an increase in air temperature, particularly mean minimum daily temperature (Oke 1982). Largely, UHI is caused by the high heat retention by the built environment, a loss of reflective surfaces (e.g., leaves) and evapotranspirative cooling by vegetation, and greater trapping of reflected radiation by elevated aerosol concentrations in the urban atmosphere (Arnfield 2003). Temperature differences between urban and rural areas are most pronounced in the first few hours after sunset as built structures slowly release their trapped heat. As a result, urban areas in temperate climates can experience fewer frost events (e.g., Melbourne, Australia; Parris and Hazell 2005). Daytime temperatures are more complex. Urban areas can be as much as 10 °C warmer in cities than rural land (Kim 1992), but temperatures can also be lower in urban areas; mesic plantings, especially in arid climates, can create an urban cooling effect by increasing evapotranspiration in the city (Brazel et al. 2000).

Water, from irrigation or precipitation, is an important control on urban climate. Precipitation can be affected by urbanization due to the increased levels of atmospheric aerosols which facilitate cloud droplet formation (Shepherd et al. 2010). As a result, precipitation can be greater downwind from urban centers (Shepherd et al. 2010). However, the relationship between urban aerosols and precipitation is not clearly understood and is strongly influenced by regional context (Schultz et al. 2007). Perhaps more important to wildlife, irrigation water inputs increase the effective precipitation rate in cities. Through irrigation, water can be available year-round, even in arid climates. Parris and Hazell (2005) found warmer urban temperatures and the use of irrigation allowed the grey-headed flying fox to establish home populations in Melbourne—a temperate, semi-arid city outside its climate range.

4.4 Water

Water is as essential for wildlife in cities as in any other ecosystem. It provides multiple urban ecosystem services, but it also collects and transports matter from and into the urban landscape. Overall, hydrologic cycles are fundamentally altered by urbanization through highly engineered infrastructure and human activity. Humans supplement precipitation inputs through outdoor water use, principally the irrigation of landscaping. This becomes increasingly important to urban hydrology when precipitation inputs limit vegetation growth. For example, in arid climates, differences in outdoor water use can create patches of vastly differing moisture conditions—from desert-like to mesic environments. Collectively, the spatial and temporal distribution of water can be dramatically different in urban systems relative to rural counterparts. Changes to the gray infrastructure also have important implications for the movement of water in and around cities. Impervious surfaces, sloped grades, and piped networks quickly drain water off the urban landscape (Pickett et al. 2011; Kaushal and Belt 2012). The increased hydrologic connectivity from urban development often negatively impacts stream and riparian ecosystems (Meyer et al. 2005; Stander and Ehrenfeld 2010). The cumulative effects of urban hydrology on streams have been termed “urban stream syndrome.” Typical features of urban streams include: incised channel morphology (Booth 1990), flashy hydrographs, high sediment loads (Trimble 1997), decreased biotic richness (Walsh et al. 2009), and broad changes to ecosystem function (Meyer et al. 2005). Urban waterways can also be enriched in nutrients, metals, and pesticides (Paul and Meyer 2008) as well as road salt (Kaushal et al. 2005). The salinity of northeast surface waters is increasing at an alarming rate and projected to become toxic to freshwater wildlife in the next century if current trends persists (Kaushal et al. 2005).

Many effects of urban hydrologic changes have been observed for wildlife populations. In fish, communities have become more similar to one another, or homogenized (Walters et al. 2003). Roy et al. (2009) found that between 22 and 66 % of the variation in urban and suburban fish assemblages could be explained by hydrologic variables alone, especially “flashiness”—abrupt changes in flow due to increases in impervious cover. Urbanization has also been linked to declines in native amphibian populations and an overall loss of diversity (Riley et al. 2005) as well as changes to macroinvertebrate composition in streams (Walsh 2004).

Urban riparian zones—potential hotspots of biogeochemical cycling—can mitigate some of the effects of urbanization by filtering toxins, sediment, and nutrients. However, urban riparian zones are often impaired by changes to hydrology (Ehrenfeld and Stander 2010). Most significantly, urban riparian zones can become disconnected from the stream and groundwater supply due to stream channel incision and decreased groundwater flow due to reduced water infiltration in urban upland soils (Groffman et al. 2003). The altered hydrology and urban riparian zone also affect urban stream and riparian function as wildlife habitat, principally through effects on stream geomorphology and riparian plant structure (Ehrenfeld and Stander 2010). For example, when riparian vegetation is lost due to drier soils, so is the capacity for that vegetation to absorb heat, resulting in higher stream temperatures (Nelson and Palmer 2007; Paul and Meyer 2008).

The many changes to upland, riparian, and stream dynamics can be organized conceptually using traditional frameworks adapted for urban systems. River or watershed approaches, specifically, have been a useful organizing tool for studying hydrology. Kaushal and Belt (2012) synthesized the major features of the hydrology of cities in the Urban Water Continuum framework. This was adapted from a framework widely used in nonurban areas called the river continuum concept (Vannote et al. 1980) that describes the structure and function of rivers from their headwaters to higher order streams. The Urban Water Continuum recognizes six key elements of urban hydrology: (1) urban infrastructure replaces first order streams, (2) nutrient retention is changed in urban headwaters, (3) there are downstream pulses in material and energy, (4) the piped infrastructure and groundwater interact, (5) hydrologic residence times affect the transfer of energy and matter, and (6) changes to land use affect biogeochemical cycles (Kaushal and Belt 2012).

Overall, the quick conveyance of water off the urban landscape has been identified as the main threat to water quality and habitat degradation. In response, cities are moving away from highly constructed systems to a system that relies more on smart design and ecosystem services. Efforts have been focused on breaking the direct conveyance of water between impervious surfaces and streams (Walsh 2004; Kaushal and Belt 2012). This includes stormwater management controls—such as bioretention, permeable pavement, and constructed wetlands—that slow runoff and increase connectivity between groundwater and soil (Collins et al. 2010). Constructed wetlands, by adding or restoring wetlands to the urban landscape, may provide a valuable wildlife habitat (Chap. 16).

4.5 Soils

Urban soils, like their nonurban counterparts, serve as the brown infrastructure of cities (Pouyat et al. 2007, 2010) and the basis for important ecological processes (Brady and Weil 2007). Urban soils can directly provide habitat for urban wildlife and indirectly support vegetation that serves as habitat and food for urban wildlife. The heterogeneity of urbanization associated with land cover is reflected in the soil; natural and social processes create a soil mosaic that is "patchy"—demonstrating high variability in structure and function (Pickett and Cadenasso 2009). Often, the structure of urban soils is altered in ways that affect ecosystem function (Pouyat et al. 2010). For example, soil compaction affects the ability of soil to absorb water and filter nutrients and pollutants. Urban soils are moved, amended, eroded, and sealed during the development and expansion of cities. Compared to nonurban soils, they often exhibit differences in pH, organic carbon and nutrients, contaminants, bulk density, soil temperature, soil moisture, and age (Lehmann and Stahr 2007). Despite this, urban soils can still serve as sinks for pollutants and nutrients, reduce storm water runoff, provide a medium for plant growth, including food production, and support habitat for wildlife.

In addition to the general changes in urban soil structure, two common properties of urban soils are likely to be of significance for wildlife: contamination and compaction. Commonly, urban soils have elevated pollutant levels. Soil contamination is a result of contemporary and historical human activities, both indirectly through activity associated with industry, commerce, and transportation, and directly through application of pesticides and soil amendments (De Kimpe and Morel 2000; Pickett and Cadenasso 2009; Schwarz et al. 2013). Contamination of urban soils is not restricted to locations with a legacy of pollution. Pollutants—metals in particular—are dispersed throughout urban areas through atmospheric transport and deposition (see *Urban Atmosphere*). As a result, even natural ecosystem remnants in and near urban areas have high levels of contamination (Pouyat and McDonnell 1991). More common heavy metals found in urban soils include Pb, Zn, and Cu due to their link with automobile use (Monaci et al. 2000) and Ar due to its link with pressure-treated wood (Heiger-Bernays et al. 2009). In addition to metals, urban soils may be enriched in polycyclic aromatic hydrocarbons (PAHs; Bradley et al. 1994) and PCBs, both of which can accumulate in soil biota (Krauss et al. 2000). Industrial zones may have elevated levels of tetrachloroethene, chromium, dioxins, sulfur, molybdenum, persistent bioaccumulative toxins (PBTs), benzene, toluene, xylene, and ethyl benzene (Heinegg et al. 2000).

Polluted soils can have important implications for urban wildlife populations. Notably, contaminants can bioaccumulate in soil-dwelling invertebrates (Kennette et al. 2002) and be transferred through the food chain (Stansley et al. 2001). Scheifler et al. (2006) demonstrated that Pb was bioaccumulating in the feathers and blood of urban blackbirds (*Turdus merula*)—and in the earthworms they ate—at levels of potential consequence to wildlife health. Many urban mammals as well contain high levels of heavy metals in their tissue (e.g., Komarnicki 2000; Dip et al. 2001). Heavy metals in urban soils are also washed into aquatic systems where plankton, fish, and other aquatic species are able to bioaccumulate heavy metals in the food chain (Chen et al. 2000). In stormwater detention ponds around Orlando, Florida, fish—a prey species for wading birds—were accumulating high levels of multiple heavy metals (Campbell 1994). Apart from bioaccumulation, some metals (e.g., Cd, Se) associated with urbanization are toxic to aquatic organisms.

Compaction and sealing of urban soils with impermeable materials is common and has consequences for multiple ecological functions. For example, reductions of particle solids and pore space in the soil (bulk density), from soil compaction (Lehmann and Stahr 2007), can reduce the infiltration capacity of urban soils (Pitt et al. 2008). Bulk density is important for individual plant needs as well as the movement of water, nutrients, and pollutants in soil. In severe cases, urban soils can exhibit imperviousness comparable to hard cover surfaces such as concrete and pavement (Gregory et al. 2006). Soil compaction is most likely to affect soil-dwelling wildlife species. For example, carabid beetle diversity was correlated with the level of soil compaction in abandoned urban lots (Small et al. 2002). Soil compaction can also affect a broader range of wildlife species through its effect on plants; the plant species present and their growth are restricted in compacted soils.

4.6 Integration of Drivers Across Space and Time

Considering abiotic drivers of urban systems in isolation limits our ability to provide meaningful synthesis and apply that knowledge to wildlife ecology. Our knowledge of ecology is robust with theoretical frameworks for understanding abiotic and biotic environment links. However, in urban ecosystems, humans act as important drivers of ecological change—altering biotic and abiotic factors and greatly increasing ecological complexity. All sets of interacting drivers—abiotic, biotic and human—are dynamic, multi-faceted, and act on multiple scales. This complexity requires that we integrate across abiotic, biotic, and human drivers of ecosystem change, as well as consider change over space and time (Chap. 3).

Conclusion

The overarching change brought about by the process of urbanization can be described by changes to urban structure. These structural changes include altered gray, green, and brown infrastructure of cities. Changes to urban structure can both directly and indirectly affect urban wildlife. For example, roads can change the movement and distribution of wildlife in cities as well as alter the composition, fragmentation, and establishment of the green infrastructure upon which urban wildlife depends (Forman and Alexander 1998). They also absorb heat contributing to a unique urban climate. Changes to urban structure drive changes to the water cycles in cities by directing water off the land, changing both the quality and quantity of water in the urban environment. Urban soils are altered by compaction and sealing as well as nutrients and pollutants that are transported along altered hydrologic cycles. Many of the changes brought about by urbanization have negative consequences for urban wildlife—degraded habitat, altered feeding and reproduction regimes, and reduced biodiversity. However, not all changes to urban structure create challenges to urban wildlife. One major key to understanding how urban wildlife will respond to the urban environment is to understand changes to the abiotic drivers of urban systems, how they vary over time and space, and how their dynamics can be used to predict future change.

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Chapter 5
Drivers of Vegetation Species Diversity and Composition in Urban Ecosystems

Anna L. Johnson and Christopher M. Swan

Keywords Vegetation structure · Habitat invasibility · Urbanization gradient · Human landscaping preferences · Neighborhood effect · Habitat fragmentation · Wildlife habitat

5.1 Introduction: The Role of Plants in Terrestrial Ecosystems

In terrestrial systems, plants are the base of the food web, and their population growth is generally limited by resource availability rather than by higher trophic levels (Hairston et al. 1960). Plants are also important components of the biogeochemical cycles that drive the movement of energy and resources. As primary producers, plants make energy available to higher trophic levels by converting sunlight, via photosynthesis, into biomass. Plants also create highly textured and structured habitat that provides food and cover for wildlife. Patterns of vegetation are shaped by variation in climate, soils, and disturbances, and provide the environmental template that drives patterns of species interactions throughout food webs. Where in this picture do urban areas fit?

5.2 Vegetation in Terrestrial Urban Systems

Ecologists increasingly recognize the pervasive impacts of human activities on the diversity, structure, and function of vegetative communities, as well as their ecosystem services (e.g. Turner 2010). With the majority of the world's growing human population living in cities (Chap. 1, Grimm et al. 2008), rapid urbanization has led to the realization that built environments must provide a diverse suite of ecological functions for both people and wildlife. Today, research foci have shifted from

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A. L. Johnson (✉) · C. M. Swan
Department of Geography & Environmental Systems,
University of Maryland, Baltimore County, Baltimore, MD 21250, USA
e-mail: annaj1@umbc.edu

C. M. Swan
e-mail: cmswan@umbc.edu

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whether substantial biodiversity exists in cities to instead what the specific mechanisms are that determine the abundance and functional role of biodiversity within cities (e.g., Blaustein 2013).

Urbanization is a combination of many processes, not all of which vary similarly across the landscape, and thus it is not clear to what extent the ecological theory developed in less human-impacted systems applies to urban systems. Decomposing urbanization gradients into the component parts may help to identify specific drivers of vegetation patterns. A mechanistic understanding of urban vegetation patterns is necessary for effectively managing urban areas, and for predicting the consequences of changes in urban vegetation for other trophic levels and for ecosystem functioning. In this chapter, we briefly review research on urban vegetation structure and function and recommend future research directions.

5.2.1 Relationship of Vegetation to the Abiotic Environment: Plant–Soil Interactions

Plant–soil interactions are tightly coupled, and urbanization can influence these relationships. The diversity, productivity, and composition of plant species can be significantly altered by soil microbial communities and related ecosystem functions, such as carbon and nitrogen cycling (Zak et al. 2003; van der Heijden et al. 2008). One consistent effect of urbanization is an increase in the heterogeneity of urban soils, due to direct (e.g., urban infill at construction sites or lawn nutrient amendments), as well as indirect human impacts (e.g., changes in microclimatic, water availability, amount of stormwater runoff, heavy metals, and other pollutants) (Pavao-Zuckerman 2008). Understanding the relationship between heterogeneous urban soils and plant communities requires an understanding of the relative importance of contemporary and local plant–soil interactions, compared to the importance of longer-term and broader-scale phenomena. These broader-scale phenomena include legacies of past land-use or the overall level of urban development in the region surrounding vegetative patches.

The results of empirical studies of plant–soil interactions in urban systems have been mixed. For example, a study of the relative importance of land-use and vegetation structure for litter decomposition in urban soils found that land-use (landfill or urban park) was more important than current vegetation composition for determining rates of decomposition, although the diversity of the vegetation had a significant effect on the abundance of soil biota (Vauramo and Setälä 2011). However, Elgersma et al. (2011) showed that 2 years after manipulating vegetation in forest patches containing invasive plants, the historical abundance of invasive vegetation had a stronger effect than contemporary vegetative community on microbial community structure and function. Heneghan et al. (2006) found similar legacy effects on soils, caused by the invasive shrub *Rhamnus cathartica.* Groffman et al. (2006) determined that forest productivity and nitrogen cycling rates were more strongly related to soil type than to the relative amounts of urban land cover surrounding forest patches, but that forest species composition was more strongly associated with the amount of surrounding urban land cover. Future research should explicate the complex interactions between urban vegetation and the abiotic environment. This type of research will improve our ability to design and manage urban green spaces that provide for humans and wildlife alike.

5.2.2 Relationship to Other Trophic Levels

Plant communities in urban environments form a "template" for other functional groups of species. These communities are assembled through a combination of natural processes and direct human manipulation (e.g., gardening). Thus, while humans generally do not directly manipulate other groups of species in the urban environment, they do have an indirect influence on their diversity and abundance through alterations to the vegetation.

For example, the potential for small-scale urban agriculture to provide food security in low-income neighborhoods has led to interest, from both researchers and practitioners, in urban pollinator communities. Pollinators are necessary for successful production of many agricultural crops, and the extent to which urban environments support a pollinator community is strongly affected by vegetation structure and diversity. Hennig and Ghazoul (2011, 2012), in studies conducted in Zürich, found increased floral diversity and abundance increased rates of pollinator visitation to focal patches throughout the city. However, the effect of local plant diversity was strongly mediated by landscape attributes such as the amount of habitat fragmentation and the proportion of green space surrounding the patch. Thus, increasing the availability of local floral resources will not always lead to increased wildlife and ecosystem services (e.g., pollination; Matteson and Langellotto 2011), and management activities aimed at increasing the availability of floral resources for wildlife may need to consider the regional context of local habitat patches (Goddard et al. 2010, see Chaps. 12, 14).

Urban vegetation also provides valuable food resources for wildlife. The regional distribution of vegetation patches in urban environments mediates the importance of local habitat for arthropods (Sattler et al. 2010; Vergnes et al. 2012), birds (Marzluff and Ewing 2001; White et al. 2005; Litteral and Wu 2012) and mammals (FitzGibbon et al. 2007; Gomes et al. 2011; Hale et al. 2012), all of which use plants for food and cover. Species respond to patch connectivity and size differently depending on their functional characteristics (e.g Sattler et al. 2010; Litteral and Wu 2012). There is not a simple formula for predicting the effectiveness of manipulation of urban vegetation for supporting desired wildlife, because the spatial scales at which plant diversity and abundance most strongly influence other trophic levels varies by taxa.

5.2.3 Relationship to Humans

Urban vegetation is important in human systems (Chaps. 3, 6), and human activities are another important driver of urban vegetation dynamics. People value the nature they experience "in their own backyards"; their every-day experiences close to home may change the way they perceive the environment, and can positively affect their relationship to nature (Miller 2006; Fuller et al. 2007). Accordingly, people select plant communities for particular locations and purposes based on the traits they possess and their ability to fulfill desired objectives. For example, street trees are selected based on their ability to tolerate local site conditions or based on aesthetic preferences. Sometimes, human preferences for a particular type of landscape can lead to problematic ecological results, such as the devastating loss of millions of American elm (*Ulmus americana*) trees to the introduced Dutch Elm Disease. American elm was one of the most popular and widely-planted species of street tree in the early twentieth century, and their broad-scale decimation led to increased awareness of the potential for heavy impacts of nonnative pest species on urban monocultures and a push to increase the diversity of urban tree plantings (Raupp et al. 2006). By increasing the diversity of street trees, urban arborists can increase community resilience to disturbance and promote wildlife diversity. Urban foresters also balance a need for diversity with human aesthetic preferences and species' ecological tolerances. For example, they select for species that require minimal maintenance, tolerate a variety of urban stressors (e.g., high-soil compaction, low oxygen available to roots, drought, and salt sensitivity), and grow to a size appropriate for their location in the landscape (Bassuk et al. 2009).

5.3 Vegetation Structure in the Urban Environment

Colonization of urban areas by both native species and escaped ornamental plants shapes urban plant diversity in unique ways. The diversity and characteristics of plant species that can colonize a site strongly influence how species are sorted, successional trajectories, and community response to disturbances. Unlike other taxonomic groups, overall plant species richness generally increases in urban areas, compared to species pools in surrounding rural landscapes (McDonnell and Hahs 2008). This increase in species richness is attributed primarily to the importation and distribution of ornamental plant species, many of which are not of native origin (Hope et al. 2003; Walker et al. 2009; Marco et al. 2010). Many of these ornamental species escape cultivation and become semi-naturalized or invasive (Kowarik 2003; Colautti et al. 2006). The purposeful introduction of nonnative plant species combined with unintended extirpation of plant species intolerant of environmental conditions in urban areas shifts the composition of urban regional species pools relative to less urbanized areas by increasing in the relative abundance of invasive, nonnative species (Williams et al. 2009).

Table 5.1 Summary of empirical results from five studies that have measured shifts in plant functional traits along urban-rural gradients. Data are shown for all traits examined by at least two of the studies. Under "Urban Abundance," the number of + reflects the number of studies documenting a change in trait abundance/proportion. The relevant citations are listed for each trait

Trait	Urban abundance			Citations
	Increase	Decrease	No change	
Dispersal vector				
Wind		++	++	Knapp et al. 2008, 2012; Thompson et al. 2008; Burton et al. 2009
Animal	+++		++	Knapp et al. 2008, 2012; Thompson et al. 2008; Burton et al. 2009
Human	++			Knapp et al. 2008, 2012; Burton et al. 2009
Water	+++	++		Knapp et al. 2008, 2012; Thompson et al. 2008; Burton et al. 2009
Seed weight		++	+	Thompson et al. 2008; Vallet et al. 2010; Knapp et al. 2012
Pollination vector				
Insect	+		++	Burton et al. 2009; Vallet et al. 2010; Knapp et al. 2012
Wind		+++	+	Knapp et al. 2008, 2012; Thompson et al. 2008; Burton et al. 2009
Self	++		+	Knapp et al. 2008, 2012; Thompson et al. 2008; Burton et al. 2009
Life duration				
Annual	+++			Knapp et al. 2008, 2012; Vallet et al. 2010
Biennial	++		+	Knapp et al. 2008, 2012; Vallet et al. 2010
Perennial	+	++		Knapp et al. 2008, 2012; Vallet et al. 2010
Leaf longevity				
Evergreen		+	+	Burton et al. 2009; Knapp et al. 2012
Deciduous		+	+	Burton et al. 2009; Knapp et al. 2012
Specific leaf area	+++			Knapp et al. 2008; Thompson et al. 2008; Vallet et al. 2010
Plant height	+		+	Thompson et al. 2008; Vallet et al. 2010
Vegetative reproduction			++	Knapp et al. 2008; Thompson et al. 2008

Plant species that are successful in urban environments tend to share a similar suite of traits, tolerances, and life-histories (Table 5.1). For example, species that are wind-pollinated, have propagules that are wind-dispersed, are evergreen, have a low specific leaf area, and are perennial occur less frequently in urban environments than in surrounding, less developed areas. However, further work is needed to empirically determine relationships between functional traits and plant responses to the environmental changes caused by urbanization. Increasingly, phylogenetic and functional trait approaches are used to describe not just the number of species in a region, but their relationships and functional roles. Incorporating these descriptions into assessments of urban biodiversity has led to the documentation of a common pattern of phylogenetic and functional homogenization of urban plant communities across the globe (McKinney 2006; Ricotta et al. 2009; Shochat et al. 2010; Knapp et al. 2012). Functional diversity, in particular, has been highlighted as an important metric for generalizing patterns of biodiversity and comparing between study sites (McGill et al. 2006). This is because functional traits, even those that are loosely linked to physiological functioning in plants, still can be valuable for describing how plant communities are sorted along environmental gradients of stress or disturbance (Grime 1977; Weiher et al. 1999; Lavorel and Garnier 2002). For example, specific leaf area is often used as a proxy for the more difficult to measure traits such as relative growth rate, palatability, or plant reaction norms (Weiher et al. 1999).

5.3.1 Plant Invasions in Urban Environments

The proportion of nonnative plant species is consistently greater in urban environments, compared to rural areas (Winter et al. 2009). One explanation for the high rate of urban invasions is the theory of increased propagule pressure, defined as a combination of the number of individuals released into a location and the number of times they are introduced (Lockwood et al. 2005). The propagule pressure model suggests that humans increase the probability of a species being invasive by repeatedly introducing it (Colautti et al. 2006).

In addition to the propagule pressure model, two additional concepts are useful for predicting and explaining likelihood of invasion: species invasiveness and habitat invasibility. Habitat invasibility refers to the abiotic or biotic community conditions that make successful invasion more likely to occur at a location. A change in disturbance regime or low diversity of the resident community, which leaves niche space available, are two conditions that often increase invasibility (Funk et al. 2008). The abiotic environment in urban areas may be more easily invaded, due to increased nutrient availability, human-aided transport of materials and propagules across the landscape, and the fragmentation of habitat. Habitat edges, which increase in proportion to habitat interiors as fragmentation increases, are zones of contact between contrasting patches, often share characteristics of both bordering patches, and can alter the rate of flux of species and materials, compared to habitat interiors (Cadenasso et al. 2003; Minor et al. 2009). For example, plants with wind-dispersed seeds tend to be less common in urban areas than in surrounding rural landscapes, because fragmentation of habitat makes it more difficult for seeds to arrive in favorable patches for germination (Cheptou et al. 2008; Knapp et al. 2012). Small, fragmented habitat patches may experience higher levels of disturbance as well, although changes in disturbance regimes are no longer considered the primary factor for predicting habitat invasibility (Moles et al. 2012).

On the other hand, species invasiveness refers to the likelihood that a particular species, with a particular suite of traits, is expected to become invasive. Traits that might predict invasiveness include polyploidy (having multiple copies of chromosomes; te Beest et al. 2011), low seed mass and short juvenile periods (Rejmanek and Richardson 1996), and self-compatibility (not needing pollen from another individual to reproduce sexually; Hao et al. 2010). In addition, human selection for desirable ornamental characteristics, such as attractive berries, may inadvertently increase urban species' potential invasiveness (Kitajima et al. 2006).

Traits that correlate with species invasiveness often are combined with models of habitat invasibility to predict areas of the landscape that may be particularly vulnerable to invasion. For example, species with life history traits less similar to species in the extant community may be more likely to invade a particular habitat patch (Funk et al. 2008; MacDougall et al. 2009). Overall, a more synthetic approach, incorporating multiple interacting processes, is necessary to explain, predict, and manage complex patterns of invasions into urban systems. Regardless of the specific causes, the consequential shifts in plant community can lead to shifts in community function (Hillebrand 2008; Raupp et al. 2010; Rodewald 2011; Eviner et al. 2012). Ultimately, reducing the diversity of functional traits present in a community may lead to reduced community resilience (i.e., the ability of a community to return to its original, pre-disturbance state) in the face of changing environmental conditions, by narrowing the range of environmental conditions under which the community is pre-adapted to succeed (i.e., the "insurance hypothesis," Yachi and Loreau 1999; MacDougall et al. 2013). For example, more diverse communities have experimentally been shown to recover more quickly after a disturbance, such as drought, than less diverse communities, regardless of the initial biomass of the communities (Van Ruijven and Berendse 2010). Also, communities in which nondominant species were experimentally removed were less drought resistant and also less resilient than communities in which both dominant and nondominant species were allowed to coexist (Mariotte et al. 2013)

5.3.2 *Human Preferences for Urban Plant Assemblages*

Whereas typical environmental variables tend to be similar in patches that are closer together in space (e.g., patterns of rock formations influence where water is available), urban land use is arranged by human design, and often decoupled from the underlying environmental features (Chap. 3). Studies conducted in Phoenix, Arizona residential yards, for example, showed strong patterns of plant community composition associated with human neighborhood-scale preferences for xeric or mesic landscapes (Walker et al. 2009). Many other studies have found "neighborhood effects" for vegetation structure and composition (Zmyslony and Gagnon 1998; Martin et al. 2004; Warren et al. 2008). Nassauer et al. (2009) tested whether homeowners were more likely to prefer yards that conformed to broad cultural norms (e.g., a well-cut lawn) or yards that matched immediately neighboring lawns, and determined that the local neighborhood effect was much stronger than broad cultural preferences. Similarly, Peterson et al. (2012) determined through surveys that while most homeowners preferred a yard that contained 50 % native plants, they assumed that their neighbors preferred turf grass dominated yard designs, and their landscaping choices were most influenced by incorrect assumptions about neighbors' preferences. Hope et al. (2003) found that plant species diversity was correlated with socioeconomic variables, as they varied across a city. Wealthier neighborhoods tended to have greater species diversity than lower income neighborhoods, a pattern that they termed the "luxury effect." Complicating the relationship between socioeconomic status and diversity, however, is the legacy of past land-uses and the age of neighborhoods, which can interact with present conditions to influence relationships between neighborhood variables of interest (Luck et al. 2009; Lowry et al. 2011). In a study of neighborhoods in Baltimore, Maryland, tree canopy coverage increased with age of houses in neighborhoods (Grove et al. 2006). Martin et al. (2004) discovered the opposite pattern in Phoenix, Arizona, as vegetation diversity decreased with increasing time since neighborhood development. These contrasting findings may reflect differences in the way in which cities are built, and demonstrate that careful consideration of the spatial and temporal patterns of urban development may be just as important for understanding the structure of urban vegetation as environmental predictors such as soil quality, water availability, and habitat connectivity.

Within neighborhoods, variation in individual homeowner choices and management strategies can influence local plant diversity. Private gardens or backyards have been estimated to cover from 16 to 36 % of total urban areas (Goddard et al. 2010); thus, the role of variation in individual preferences for particular types of landscapes may play a significant role in determining city-wide patterns of species composition and diversity. The frequency with which homeowners undertake particular landscaping activities has been shown to have a significant relationship to characteristics of vegetative composition and diversity (Loram et al. 2011). A recent study of people's preferences for particular plant traits and the composition of their garden flora showed that these preferences can drive significant variation in garden flora composition (Kendal et al. 2012). Thus, human preferences (e.g., for plants with evergreen leaves and large, colorful fruits) may be a strong selective force in urban environments, alongside changes in environmental variables and landscape connectivity (Williams et al. 2009).

5.3.3 *Heterogeneity in Urban Plant Communities in Space and Time*

The majority of studies of spatial patterns of urban biodiversity have used an urban-rural gradient approach (McDonnell and Hahs 2008). The "urbanization" gradient is made up of a combination of many different environmental gradients, not all of which are changing at the same rate or in the same direction. Thus, using particular "urbanization" measures, such as percentage of impervious surfaces or population density, may obscure other environmental variables that are changing in contradictory manners. Even so, general patterns have emerged in the literature, including an increase in native plant species extirpation with urbanization (Williams et al. 2005).

The intermediate disturbance hypothesis has been one of the primary models used to interpret patterns of urban plant diversity (Connell 1978). This model predicts that the highest levels of species coexistence in a system will occur at intermediate disturbance frequency, magnitude, or time since a disturbance. The theory has been applied to explore the coexistence of native and nonnative species along urban-rural gradients, or within the urban environment, between patches that vary in disturbance level (e.g., Porter et al. 2001; Mandryk and Wein 2006; Catford et al. 2012). The expectation is that species diversity will be maximized at intermediate locations, where both native and invasive species occur in the same communities, in relatively even proportions. These gradients of disturbance, however, may interact with patterns of urban land use and neighborhood characteristics, and obscure expected relationships (Porter et al. 2001).

Another commonly used framework for understanding patterns of urban plant diversity is borrowed from island biogeography theory. Island biogeography is a quantitative predictive model of a dynamic equilibrium of local species richness set solely by island size and distance from the source pool of immigrants (MacArthur and Wilson 1963). Island biogeography's core theories have been applied to help understand how habitat fragmentation and alteration in urban areas may change patch-level and regional-level species diversity. For example, Honnay et al. (1999) found that smaller and more isolated forest patches in a suburban matrix harbored fewer plant species. A study of vacant lot plant diversity showed that older vacant lots had greater plant diversity, and that diversity levels seemed to approach equilibrium after 30 months (Crowe 1979).

Metacommunity theory is a relatively recently developed framework that is beginning to be applied to patterns of urban biodiversity. A metacommunity is conceived as multiple local species assemblages structured by local environmental constraints and biological interactions and connected by dispersal between patches (Leibold et al. 2004; Cottenie and De Meester 2004; Urban 2004; Chase 2005; Stevens 2006). Species interact and are assembled along local environmental gradients, but are connected via dispersal to a broader, common regional species pool. Metacommunity theory incorporates elements of other earlier frameworks, especially that of island biogeography. In urban ecosystems, human activities not only fundamentally alter environmental gradients and the strength and direction of interspecific interactions, but also the broader-scale movement and displacement of species, due to human management choices and land use patterns.

Fragmented landscapes reduce connectivity between ecological communities. A reduction in dispersal is generally thought to reduce regional-scale population persistence, and may also lead to a loss of genetic diversity as populations become isolated. In urban ecosystems, connectivity may be constrained by dispersal of organisms through corridors like parks, riparian zones, and median strips. The result of a loss of connectivity can be a reduction in local species diversity, or an increase locally in the relative abundance of species exhibiting greater dispersal ability, and which are less affected by reductions in habitat connectivity. However, human actions may effectively enhance dispersal for particular species, and counteract expected losses in individual abundances or population diversity. This may occur directly via planting efforts, as in the context of landscaping, restoration, and gardening, or indirectly via the spread of nonnative species (e.g., as a contaminant in seed or soil mixes).

Variation in homeowner preferences for plant species composition is likely to contribute more to regional plant diversity than simply the number of species people harbor at a particular location. A survey of urban garden biodiversity patterns in Sheffield, UK demonstrated that although the local diversity of urban gardens did not vary significantly from the local diversity of areas termed "semi-natural" and "urban derelict" land, the regional diversity of all the urban gardens surveyed in the study was more than twice as high as that found in any other type of plot surveyed, including pasture and woodlands on either acidic or alkaline soils (Thompson et al. 2003). Spatial heterogeneity in environmental conditions in urban environments, coupled with variation in human actions at multiple organization levels (e.g., residential, neighborhood, jurisdictional) underscores the need to consider compositional turnover, in addition to levels of local diversity, to understand patterns of plant biodiversity at the scale of an entire city, and the relative contribution to overall diversity that specific portions of the urban landscape provide. For example, to plan effective citywide conservation strategies it is important to understand whether privately owned land contributes more to overall regional diversity than publicly owned land, such as city-managed parks or streetscape plantings.

Plant community composition changes through time in urban ecosystems, but the rate and distribution of this change is mediated by human activity. Ecologists often rely on successional climax theory to conceive how species composition at a location changes through time. In general, fast-growing ruderal species with the ability to disperse more widely are expected to colonize newly opened habitat. These are replaced slowly by more slow-growing species that may be more efficient at gathering and using resources. In urban ecosystems, this traditional perspective may not apply broadly to explain the composition of differently-aged communities. The urban landscape is extensively disaggregated, with a high degree of heterogeneity in human influence on ecological organization. This spatial heterogeneity does not remain static, but changes over time.

Shifts in patch-level land use can reset the colonization process in three ways. First, in some cases all plants are removed and no attempt is made to establish a community (e.g., following demolition of a building structure, if land is left vacant as in the case of urban vacant lots or brownfields, Fig. 5.1, left). The colonization process proceeds with species composition developing as a consequence of local habitat filters and the availability of colonists from the regional species pool. In such cases, one often observes dominance of a few species relative to nonurbanized environments, as well as high proportions of nonnative species.

Second, management or use of land with an already established plant community may simply stop. In this case the colonization process is not reset, and the initial species composition of the community initiates the trajectory of post-abandonment species assembly (Fig. 5.1, center, as in secondary-succession). Lack of human management may result in the decline of species that require care to persist in the

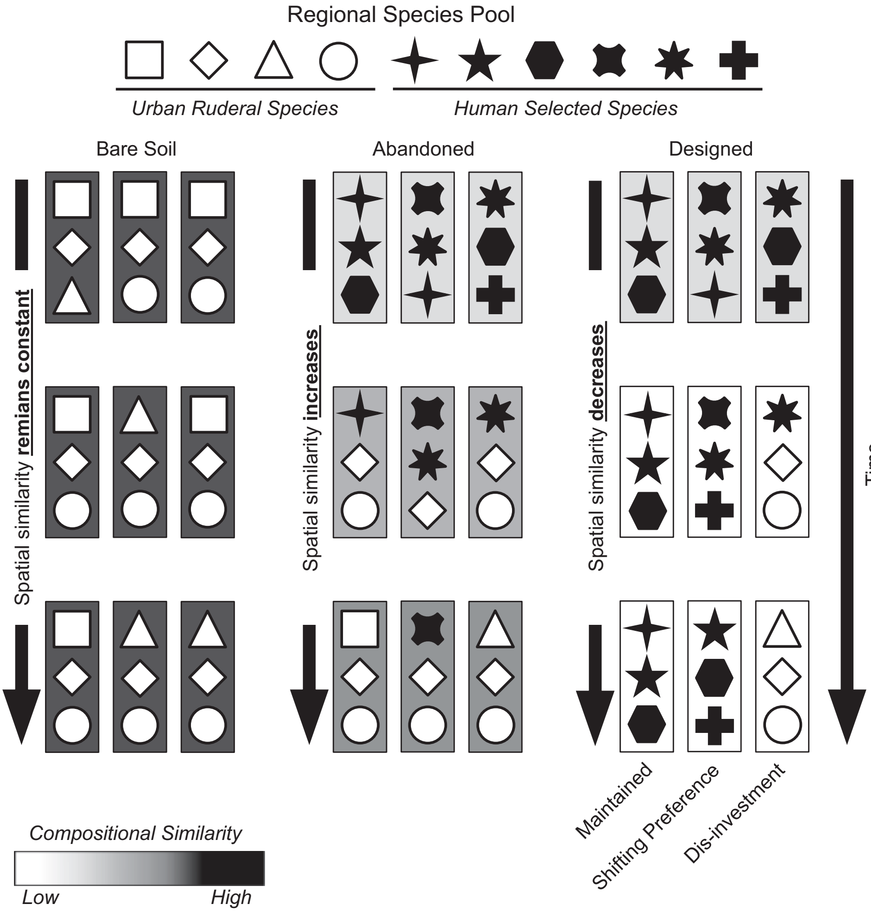

Fig. 5.1 Three scenarios illustrating the shifts in community composition in urban ecosystems. Each symbol represents a unique species or functional group. Sets of three boxes represent a metacommunity of three patches of similar initial type. The degree of shading reflects the magnitude in compositional similarity among habitat patches within each metacommunity. The *Bare Soil* category describes areas where vegetation was removed and sites are allowed to colonize naturally from the regional species pool. *Abandoned* areas are locations where humans once designed community composition via selective planting but then left unattended. This category could potentially consist of remnant habitat patches that have become surrounded by the urban matrix and no longer are connected to the original species pool, since supplanted by urban species. *Designed* areas, like Abandoned, initially comprise combinations of human selected species, but proceed along different assembly trajectories based on human behavior/decision-making. Under the reasonable assumption that diversity of urban ruderal species (*open symbols*) is lower than plant species chosen by humans (*closed symbols*), and that the composition of each grouping are distinct, community assembly should proceed with different compositional outcomes that depend on the initial conditions and the role humans play in facilitating coexistence. When land is completely open to colonization (Bare Soil, *left*) it is colonized predominately by urban ruderals, with constrained shifts in composition in space and time. Alternatively, where humans formally maintained coexistence by selecting species (Abandoned, *center*), ruderal species replaced human-selected species prone to local extinction without human intervention. This leads to convergence in community composition. Designed communities (*right*) may shift in one of three ways. Assemblages maintained by humans may not change at all through time. Community composition of human selected species may change over time as preferences shift for some species over others. Designed communities may cease to be maintained, and the disinvestment by humans leads to replacement with urban ruderals

urban landscapes, with replacement by the more ubiquitous urban plant species. While initial composition might vary, over time plant communities may converge in species or functional composition (e.g., Fukami et al. 2005), or remain divergent or randomly assembled with respect to the regional species pool (e.g., Schleicher et al. 2011).

In the third case, community composition is selected based on particular landscape design goals (e.g., maintained gardens, landscaped parks, or restorations Fig. 5.1, right). Of the three described, this scenario of human-influenced succession can create the greatest temporal and spatial variation in composition. Initial composition reflects a purposeful process that is constrained by human perception and valuation of space. Maintenance of composition through time does not parallel the traditional successional sequence. The level of maintenance (e.g., weeding or replacement of species as they senesce and superseding natural colonization-extinction patterns) and change through time may instead reflect the ability for individuals or organizations to make an economic investment in landscape composition (luxury effect as described in Hope et al. (2003)).

Conclusion

As of yet, few studies have monitored long-term dynamics of urban communities or used a chronosequence approach (replacing space for time) to explore how urban communities change through time, while controlling for variation in initial starting composition, land use, disturbance regimes, and environmental variables. Overall, complex patterns of spatial and temporal heterogeneity in urban plant community composition are a product of multiple drivers, including human perception of what plant species are desirable, the direct and indirect effects of human presence on local environmental conditions and shifting land use practices. Future studies of urban vegetative communities should focus on explicitly addressing these different drivers, and exploring how they vary in intensity and direction across the urban landscape.

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Chapter 6
The Urban System: Social Drivers

Charles Nilon

Keywords Social drivers · Race · Ethnicity · Income · Environmental justice

6.1 Introduction

This chapter explores how social, economic, and demographic factors influence wildlife in urban ecosystems. Recognizing that cities can be viewed as ecosystems where there is interaction among social and ecological subsystems was a key development in the management of urban ecosystems (see Chap. 3). The focus on cities as linked social and ecological systems has led to several different perspectives on how socioeconomic and demographic factors drive ecosystem processes in cities and ultimately impact wildlife conservation and management. Ecologists, managers, and others dealing with natural resources in cities bring unique approaches and perspectives to understanding social drivers of urban wildlife ecology. The goal of this chapter is to provide background information on the use of sociology in understanding the spatial form of cities, human behavior, wildlife habitat structure, and patterns of species richness.

6.2 Social Patterns in Cities: Drivers of Urban Form and Spatial Pattern

Understanding how social drivers influence urban wildlife requires an introduction to how social scientists have explained patterns and differences in cities. In the early 1900s, sociologists recognized that patterns of social and spatial inequality might shape the form and pattern of urban development. Robert Park, a sociologist at the University of Chicago, was interested in understanding how cities change over time. He viewed this change as something like ecological

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C. Nilon (✉)
Department of Fisheries and Wildlife Sciences, University of Missouri,
302 ABNR, Columbia, MO 65211, USA
e-mail: nilonc@missouri.edu

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succession, which he interpreted as a predictable pattern of change instigated by immigrants moving into a city center. Park and his colleagues stressed that social, ethnic, and economic differences created segregation in cities and that new immigrants faced the greatest amount of segregation. The pattern of segregation and waves of immigration into cities created zones in the city with unique yet predictable socioeconomic and ethnic characteristics. They proposed describing cities as a series of concentric zones surrounding the central city with orderly patterns of change as different social, economic, and ethnic groups became established and then moved progressively away from the center city (Park 1915; Park et al. 1925). Over time, researchers shifted their focus from explaining patterns of settlement to studying how social, ethnic, and economic differences led to patterns of inequality within a city (Drake and Cayton 1945). Drake and Cayton were pioneers in this field, and explored how race and segregation shaped the day-to-day lives of Black residents in Chicago. Their study of the Bronzeville area of Chicago considered how spatial patterns in the city, shaped by long-term patterns of racial segregation, in turn shaped access to jobs, participation in social organizations, and education.

Geographers in the 1950s developed the field of social areas analysis to study patterns of differentiation in cities (Shevky and Bell 1955). Social areas analysis is also being used to understand the look and feel of contemporary metropolitan areas, suggesting a link between demographic, economic, and social variables and urban form and greenspace pattern (Knox 1991). Contemporary studies using social areas analysis also use socioeconomic status as a composite scale based on family income, education, occupation, and family structure to examine a range of policy issues such as patterns in health and wellness, functional illiteracy, and unemployment (Maloney and Auffrey 2013). These two contemporary applications of social areas analysis are useful in developing the concept of social drivers of urban wildlife habitat.

6.3 Social Drivers of Urban Wildlife Habitat

Much of our knowledge of social drivers stems from research done between the 1970s and 1990s on patterns of urban vegetation in cities. These studies used social areas analysis and other approaches developed by social scientists to explain patterns in the distribution of urban canopy cover and vegetation in residential lots. These studies are important in understanding both links between socioeconomic factors in cities and greenspace pattern, and in developing an understanding of some of the variables that drive patterns of wildlife abundance and distribution in urban areas.

Schmid's (1975) intensive study of the vegetation in front yards of randomly selected neighborhoods in the Chicago, Illinois metropolitan area, related patterns in vegetation to census variables measuring years of education, income, value of owner-occupied housing, and amount of owner-occupied housing.

Schmid (1975) found that individual lots differed in vegetation structure and species composition, reflecting the influence of homeowner and resident decisions; however, he also found that lower income neighborhoods lacked lots where landowners had planted dense vegetation or allowed existing vegetation to remain. Whitney and Adams' (1980) research on street and yard trees in Akron, Ohio, used income, home value, years of education, and age of housing as independent variables predicting street and yard tree community composition. More affluent neighborhoods were characterized by large wooded lots and by what the researchers described as a mixed suburban vegetation complex. Whitney and Adams (1980) hypothesized that this suburban vegetation complex was associated with wealth and prestige, in part because maintaining wooded lots required high home-building costs. In contrast, low-income neighborhoods had smaller lots with vegetation reflecting the planting decisions of residents who lived in the house in previous decades and lots that were a mix of weedy species reflecting a lack of maintenance by property owners. Talarchek's (1985, 1990) research sought to identify demographic and socioeconomic predictors of tree species richness and vegetation pattern in New Orleans, Louisiana. He found that tree canopy cover was predicted by race, income, and poverty rate, with high percentages of tree canopy occurring in census tracts with a high percentage of White residents, high median home value, and a high percentage of single family homes. The census tracts with less tree cover and with a high percentage of cover by herbaceous vegetation were negatively associated with households with families, single family housing, and a large census tract area, an indication of large lots associated with suburban residential neighborhoods.

In the same time period as the vegetation studies, Williamson and DeGraaf (1981) compared bird communities and bird habitat structure in Washington DC's Rock Creek Park and in two neighborhoods bordering the park. Neighborhoods on the east side of the park were dominated by multi-family apartment buildings where most residents were renters. Yard vegetation was limited to street trees and scattered shrubs, attracting high densities of nonnative birds such as rock doves (*Columba livia*) and house sparrows (*Passer domesticus*). Neighborhoods to the west of the park were characterized by owner-occupied single-family housing with front yards dominated by a mix of trees, grasses, and shrubs, attracting greater densities of native species such as northern cardinals (*Cardinalis cardinalis*), blue jays (*Cyanocitta cristala*), and song sparrows (*Melospiza melodia*).

6.3.1 Demographic Drivers

The past decade has seen an emphasis on understanding social drivers by studying resident and property owner's decision making. This has expanded to broader theory focused on urban residents' lifestyle behaviors and purchasing power as drivers of social stratification that result in discrete clusters of neighborhoods in cities that are hypothesized to have different patterns of vegetation (Grove et al.

2006). Other researchers have expanded this approach to individual behavior by studying the complex factors that shape how individuals make decisions about their gardens, streets, and neighborhoods (Goddard 2010; Williams 2002; Wolford 2003). Goddard (2010) summarized much of this literature by noting that these places are “socioecological constructs” for understanding how people and their activities shape urban biodiversity. Although this approach has been criticized for an over-reliance on the role of race and class in shaping individual preferences (McFarlane 2006), these studies illustrate the value of using socioeconomic variables as measures of social stratification and as tools for understanding social drivers of ecological change.

Studies of demographic drivers have sought to move beyond describing patterns of vegetation. Richards et al. (1984) showed that social factors were not the only driver of the amount and structure of residential greenspace. In areas where residence shared similar social characteristics, there was a greater heterogeneity in amounts of residential greenspace and vegetation than on average throughout the city. Variation in shrub and ground cover and garden areas was high among individual lot features but lacked obvious patterns at large scales. Most of this variation in vegetative structure on individual lot features could be linked to individual preferences for cover and greenspace. This led Palmer (1984) and Richards et al. (1984) to develop the concept of neighborhoods as areas with discrete vegetation shaped by residents and their preferences, with those preferences shaped in part by socioeconomic status. Building on this Burch and Grove (1993) and Grove and Burch (1997) hypothesized that gender, property rights, technological change, and other variables might influence urban residents’ decisions about managing urban vegetation and in turn create patterns of difference in urban vegetation within a city. These and similar hypotheses have led to a number of studies testing relationships between proposed social drivers and measures of vegetation and wildlife species richness.

6.3.1.1 Race and Ethnicity

Studies of race and ethnicity as drivers of wildlife habitat have documented differences in preferences for vegetation and in vegetation pattern and structure between neighborhoods dominated by White residents and neighborhoods where the majority of residents are members of racial and ethnic minority groups. Peterson et al.’s (2012) study of landscaping preferences of Raleigh, NC residents found that African-American residents preferred photos of yards dominated by turf twice as often as Whites, whereas White residents preferred scenes with 50 % native plantings. Kaplan and Talbot’s (1988) study of Black and White residents of Detroit found that both groups had high regard for nearby nature and shared a preference for scenes with water, residential streets, and attractive individual trees. White residents preferred scenes with wooded areas with few buildings visible, whereas Black residents preferred open settings with a few large trees where institutional or residential areas were visible, and scenes of greenspaces with walkways and benches. Preference studies are important because they are the indicators of the kinds of wildlife habitats that residents like and manage for, ultimately influencing wildlife species composition and abundance (Grove et al. 2006).

Additional studies have compared vegetation among minority and nonminority neighborhoods. Landry and Chakraborty (2009) found that public spaces in minority neighborhoods may have fewer street trees than similar public spaces in White neighborhoods. Heynen et al. (2006) reported similar results for street trees in Milwaukee, noting that the differences in street tree abundance were in part due to fewer trees being planted and lower tree survival in African-American and Latino neighborhoods. Nilon and Huckstep (1998b) found that socioeconomic variables were correlated with access to greenspace, kinds of greenspaces in neighborhoods, and the types of disturbances that impact greenspaces adjacent to the Chicago River. Black residents with lower socioeconomic status lived adjacent to greenspaces with more trash, more dumping, more soil compaction, and more vandalism than other residents. This pattern had consequences for wildlife management. Small woodlots adjacent to both groups of neighborhoods were similar in structure and plant species composition; however, small woodlots adjacent to neighborhoods with Black residents with lower socioeconomic status had more trash, more dogs and cats, and had a different small mammal community that included nonnative Norway rats (*Rattus norvegicus*) and house mice (*Mus musculus*; Nilon and Huckstep 1998a).

An important part of understanding race and ethnicity as social drivers is recognizing the broader context shaping how people of color experience wildlife and wildlife habitats. Wolch et al.'s (2000) study of African-American women in Los Angeles documented the varied experiences that these women had with animals. Their perceptions of wildlife were shaped by: stories of a mix of first- and second-hand encounters with animals from family and friends; knowledge about wildlife and wildlife habitats from childhood experiences such as field trips and summer camps, hiking, and camping; and experiences and familiarity with common wildlife species. The broader context shaping individual behavior, preferences, and knowledge was defined by age and where the individual grew up. Younger residents who were raised in cities had very different experiences from older women from families who hunted and had more rural experiences (Wolch et al. 2000).

Van Velsor and Nilon's (2006) study of Black and Latino teenagers in Kansas City also documented the importance of varied experiences that shape perceptions of wildlife. Underlying these difference in perceptions were formative experiences in nature and supportive messages about nature from family and peers. The students who had the most positive perceptions of wildlife were those who interacted with wildlife at the youngest ages and received positive messages about wildlife and nature from family and friends.

Specific preferences for vegetation structure that are driven by race and ethnicity may also be shaped by a broader range of factors. Washington's (2008) study of the Block Beautiful Movement described the origin of neighborhood cleanup and beautification efforts in urban African-American neighborhoods starting in the 1920s. These efforts, led primarily by Black women, were viewed as tools for addressing issues of poverty and lack of services from local governments that were common in segregated neighborhoods at that time. Cleanup efforts were viewed as a tool for building sense of pride and community control among neighborhood residents, and as a showing that Black residents valued beauty and well-maintained yards and houses. A key part of the Block Beautiful Movement was an emphasis on yards with a clean, manicured appearance. Weeds, unmowed yards, and brushy areas were viewed as signs of the neglect by the resident or property owner. Washington (2008) views the Block Beautiful Movement as an environmental movement within that Black community that continues to shape perceptions and preferences for managing vegetation in residential areas. These preferences may be reflected in differences in vegetation and wildlife habitat between majority White areas of cities and areas where the majority are people of color.

6.3.1.2 Income

Recent studies of plant and bird abundance in cities have evaluated socioeconomic factors that drive wildlife habitat structure and wildlife species abundance at a neighborhood scale. These studies do not show a clear relationship between income and species richness. Melles (2005) found that the diversity and abundance of native bird species in census tracts in Vancouver, British Columbia, was positively correlated with a higher median family income and having a university degree. Kinzig et al. (2005) found that the median household income was positively correlated with the number of bird and plant species in Phoenix neighborhoods, and positively correlated with the bird species richness in small urban parks. Loss et al. (2009) studied the relationships between socioeconomic and environmental factors and bird species richness in Chicago neighborhoods, finding that per capita income was not related to the bird species richness, but was negatively correlated with the native bird species richness and positively correlated with the number of nonnative bird species. Strohbach et al. (2009) studied socioeconomic variables and their relationship to bird species richness in Leipzig, Germany, finding that residents living in neighborhoods with high socioeconomic status had high bird species richness around their homes. These neighborhoods were located near high quality greenspaces, whereas lower social status was associated with species-poor neighborhoods, which often were located on the fringe of the city.

As a social driver, income is an indirect measure of wealth or affluence. Most studies considering income and biodiversity hypothesize that affluence and wealth are tied to decisions about managing wildlife habitats at a lot or neighborhood scale. Underlying this hypothesis is an assumption that affluence provides resources that can be used in managing for wildlife or that affluence allows individuals to live in neighborhoods with habitat features that support a diverse group of wildlife species. However, results from the studies described above and work done by other researchers suggest that wealthier neighborhoods are not always neighborhoods with the best wildlife habitat and greatest number of wildlife species.

6.3.1.3 Education and Knowledge

Knowledge and information enable people to make decisions concerning the management of urban ecosystems. Kellert's (1976) study of the US public's attitudes, knowledge, and behavior toward wildlife found that residence, age, sex, race, and socioeconomic status are correlates of knowledge. Differences in these variables mean that residents of cities differ in access to knowledge and information that is relevant to making management and policy decisions; therefore, knowledge and information act as social drivers of some processes in urban ecosystems. Level of education is often used as a surrogate for knowledge, and is often viewed as a social driver influencing wildlife habitat. Kellert's (1976) work has led to a focus on years of education as a metric of knowledge and an assumption that individuals with more years of education are more likely to participate in conservation programs. Studies of urban residents who participated in conservation programs (Broun et al. 2009) and lived in conservation subdivisions (Allen et al. 2012) support this observation. However, the way in which education acts as a social driver is less clear.

Wildlife managers in cities are interested in residents' understanding of management and sustainability concepts and their ability to apply them on their properties. This interest extends to providing a variety of educational materials to urban residents with the hope that they will be used to impact habitat. Hostetler et al. (2008) and Allen et al. (2012) surveyed residents of conservation subdivisions finding that level of education, used as a surrogate for knowledge, was associated with living in the subdivision, and that education materials were successful in changing some of the management practices used by homeowners. Managers were also interested in how knowledge influences decisions about wildlife habitat made by local governments. Azerrad and Nilon (2006) surveyed planners in Washington State to identify gaps in knowledge that hindered use of guidelines for conserving wildlife habitat during development, finding that most planners lacked information on the actual species and habitats that occurred in cities and suburbs where they work.

The broader context of knowledge and how it drives decisions about management suggests that the link between knowledge and years of education is less direct. Research on the perception and knowledge of urban greenspaces, places that are important wildlife habitats, has found that people are most knowledgeable about places within 1 km of where they live, and that they are most likely to be engaged in making decisions about how these places should be managed (Kaplan and Kaplan 1989; Kaplan et al. 1998). Spirn (2005) argued landscape literacy is defined as what information local residents need to know to participate in decision making about environmental issues in their neighborhood. Lack of landscape literacy may hinder residents' abilities to participate in discussions on planning and conservation. Landscape literacy typically includes the ability to read the neighborhood landscape, understand historical factors that have shaped it, and interpret stories about its history. Spirn's efforts to teach landscape literacy incorporate the idea of a landscape as the mutual shaping of people and place and landscape literacy as the ability of people to read the stories embedded in their landscape in ways that empower them to make choices and decisions about the management of their neighborhood.

Wildlife managers in cities may wish to focus on incorporating wildlife and wildlife habitats into the concept of landscape literacy. Wildlife conservation programs that engage local residents in understanding where they live and that address wildlife issues as a part of residents' day-to-day life and experience seem much more likely to be successful than programs targeting an elite group of well-educated residents.

6.4 Culture is a Framework for Understanding Social Drivers

Social drivers are ultimately an expression of culture. Millard (2010) noted that nature shapes and informs culture at a local level, largely through the pattern of intended and unintended greenspaces, and influences how people view and manage nature. Hough (2004) noted that nature is reflected culturally as the vernacular landscape of a city, the built landscape that was developed in response to local environment. The vernacular landscape described by Hough (2004) included use of native species in landscaping, a mix of remnant natural habitats, and built and designed landscapes that often incorporated features associated with natural habitats. In contrast to the unique vernacular landscapes that have developed in response to local conditions, the globalization of design culture has resulted in the use of similar urban design and planning structures, similar landscape architectural styles, and similar plant and construction materials in cities throughout the world. This cultural emphasis on a globalized design has been a significant factor in the spread of nonnative plant species in cities, and the loss of design features that mimic natural habitats (Ignatieva and Stewart 2009; Müller et al. 2013, Chap. 14).

At the lot and neighborhood scale, culture provides the context through which people identify and define nature in cities. Much of the research on this topic has considered how people develop a sense of place or identity around where they live and how this is reflected in use and management of greenspaces near where they live. Williams' (1988) study of a Black working class neighborhood in Washington, DC, considered how class, race, and links to a southern rural heritage shape culture and influence urban residents, including gardening practices that impact wildlife habitats in yards and neighborhoods. Additional research in the Anacoastia section of Washington, DC (Williams 2001; McFadden-Resper and Williams 2006) and several St. Louis, MO neighborhoods (Wolford 2003), explored how culture shapes management of residential yards and neighborhood greenspaces.

South African cities provide unique case studies of how culture drives ecological processes. City planning between 1948 and 1994 was driven by apartheid laws resulting in communities with separate White, Colored (mixed race), Indian, and African areas. Since 1994 and the end of apartheid laws, the urban form has increasingly been shaped by patterns of poverty and inequality (Roberts 2003). Researchers at North-West University in South Africa have studied residential areas in the Tlokwe City Municipality comprised of Potchefstroom and its former segregated townships (Lubbe et al. 2010; Lubbe 2011). Segregation laws under apartheid-established patterns of social and cultural differences that have been maintained after segregation was legally abolished. Gardens in areas with more affluent White residents contained more species but with a high percentage of ornamental species, including many nonnative plants. Gardens in lower income Black areas contained fewer species but more species used for food and medicines, including a high percentage of native plants. Smith's (2004) study of breeding birds in these neighborhoods and former townships found that predominately White residential areas had greater bird species richness and abundance, suggesting that these residential areas provided sufficient nesting sites and foraging areas for a greater number of species and guilds. This research placed behavior and attitudes into a broader cultural framework shaped by access to open space and historical patterns of segregation in urban areas. The findings of these studies suggest that wildlife management at lot and neighborhood scales may be a cultural expression shaped by amounts and kinds of greenspace and preferences of local residents.

Cultural values also drive ecological process and shape the environment through the design process (Millard 2010). One of the best examples of the role of culture as a factor shaping wildlife habitat is the conditions associated with parks and cemeteries—some of the larger patches of designed landscapes in cities. Darnall (1983) described how the Romantic Movement of the eighteenth and nineteenth centuries and perceptions of the natural landscape influenced the design of parks and cemeteries in the USA, using Bellefontaine Cemetery in St. Louis as an example of a cultural influence on design. Bellefontaine Cemetery has continued to maintain an open woodland condition that was part of its original design. Azerrad and Nilon (2001) found that Bellefontaine Cemetery provided open woodland that was important in the city for red-headed woodpeckers (*Melanerpes erythrocephalus*) and other species that were rare elsewhere in the city, and stressed the link between maintaining cultural elements of the cemetery as a tool for managing habitat for less common species.

San Francisco's Golden Gate Park is another example of design practices that are both cultural elements and significant to wildlife (Cicero 1989). Ponds were built as design features in the park but had a variety of habitat features and sizes reflecting different design practices. Waterbird and shorebird abundance and species composition were linked to several design features, including shrub cover around the shoreline of larger ponds, overhanging vegetation, flooded vegetation, and small islands.

Wildlife management and ecological restoration projects in urban areas are also cultural expressions. Gobster's (2007) review of ecological restoration in Chicago's Lincoln Park found that habitat management in the park changed land uses of specific places in the park that were used by people. Some of the restoration sites were viewed by managers and restoration advocates as dangerous places associated with gangs and drug dealing. However, restoration made other sites more popular with diverse groups of users. For example, restoration of a shoreline area that was previously an informal and degraded play area for children made it popular with children as well as anglers. Gobster (2007) stressed the importance of using a community-based approach to restoration and management in cities that incorporates the values and perceptions of a wide range of residents. This community-based approach is critical to avoid management efforts being approached as an elite activity that only a few residents will appreciate.

Conclusions

This chapter summarizes recent literature on social drivers of urban ecosystem processes. Research on this topic builds on social science concepts about social, economic, and demographic factors that are correlated with where people live and with peoples' access to resources. Ecologists, land managers, social scientists, and city planners have used these social science concepts to understand and predict how residents and their activities influence urban ecosystem processes. This work started with explaining patterns in cities and has expanded to explain mechanisms that link processes to behaviors and decision making. The results of research on social drivers are important to managers because they point to the diverse ways that people shape wildlife habitats in cities and to the importance of engaging people in the design, implementation, and evaluation of management projects in cities.

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Chapter 7
Wildlife Responses to Urbanization: Patterns of Diversity and Community Structure in Built Environments

Nancy E. McIntyre

Keywords Abundance · Biodiversity · Community assembly · Competition · Diversity–productivity relationship · Dominance–diversity relationship · Intermediate disturbance hypothesis · Latitudinal · Species–area relationship · Succession · Urban

7.1 Background

As the study of biodiversity, the field of community ecology seeks to understand what factors govern the formation and maintenance of multispecies assemblages (i.e., how and why certain species coexist). The structure of communities is described by their diversity—comprised of richness (number of species present) and evenness (the relative abundance of each species)—and interactions among the constituent species. Communities are biotic components within ecosystems, and because cities can be viewed as an ecosystem (albeit one characterized by high-density human habitation and built structure; McIntyre et al. 2000), understanding how wildlife responds to the ever-increasing scope of urban development makes urban community ecology an important component of contemporary wildlife science.

Urbanization changes both the abiotic and biotic environments, altering resource dynamics, the physical environment, and climate; these changes in turn induce an alteration in species composition in urban areas (Adams 1994). As a consequence, urbanization is considered to be the leading cause of wildlife species endangerment in the USA (Czech et al. 2000; McKinney 2002).

This chapter is structured to first provide an overview of the main foci and findings of community ecology (with an emphasis on the various interspecific interactions that shape communities), followed by an examination of how urbanization

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N. E. McIntyre (✉)
Department of Biological Sciences, Texas Tech University,
Lubbock, TX 79409-3131, USA
e-mail: nancy.mcintyre@ttu.edu

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alters these patterns, and concludes with management implications for urban wildlife communities.

7.2 Fundamental Patterns Within Biotic Communities

Since community ecology emerged as an independent field of study in the latter half of the nineteenth century, several recurrent patterns have been revealed that hold true for various taxonomic groups in various locales, including urban areas (see Fig. 7.1). Foremost of these patterns is the species–area relationship, one of the most venerable in ecology, which states that larger areas tend to support more species (Fig. 7.1a). This relationship also holds true for urban areas but is complicated by the context of a locality; urban habitat remnants tend to possess lower richness than do comparably sized habitat patches located within a rural context (Chace and Walsh 2006). This may be due to decreased recolonization (effectively no replacement of a species once it is lost from a patch), particularly for urban remnants that are a considerable distance from any nonurban sources of colonists. Island biogeographic theory (MacArthur and Wilson 1967) holds that habitat patch size acts in tandem with patch isolation in determining species diversity (a balance between colonization and extinction); urban patches have been shown to conform to the predictions from island biogeographic theory (Crooks 2002).

Another fundamental community pattern is the latitudinal gradient in biodiversity, whereby high diversity is encountered at the equator and decreases as one moves towards the poles (Fig. 7.1b). There are many explanations for this pattern, and a few exceptions to it (Willig et al. 2003). A paucity of cross-city comparisons, however, means that the effects of urbanization on this pattern are poorly understood.

In terms of species abundance (i.e., dominance or its converse, evenness), rarity is common and commonness is rare—i.e., most species are represented by relatively few individuals in a community—but the converse pattern tends to hold true for urban areas. The shape and slope of the relationship between abundance plotted against the decreasing abundance rank order of species in a community (Fig. 7.1c) have been used to infer aspects of competition for and partitioning of resources (Magurran 2004). Communities in urban areas often exhibit a geometric dominance-diversity curve, a steep and right-truncated indicator of exaggerated dominance in relatively depauperate assemblages that are typically under greater abiotic than biotic control (e.g., Diefenbach and Becker 1992; Yang et al. 2011).

A fourth widely seen pattern has been termed the intermediate disturbance hypothesis (IDH) and states that diversity tends to be greatest at sites experiencing neither too much (in terms of severity or frequency) nor too little disturbance (Connell 1978; Fig. 7.1d). The unimodal pattern is generated by limitations at the extremes of the disturbance axis: at the high end of disturbance, abiotic factors limit diversity, and at the low end, biotic factors such as competitive exclusion reduce diversity. Urbanization is commonly considered an environmental disturbance (e.g., Blair and Launer 1997; Niemelä et al. 2002; Zerbe et al. 2003), yet it is in fact not a classical disturbance (e.g., like fire or flood) because people choose whether there is a return

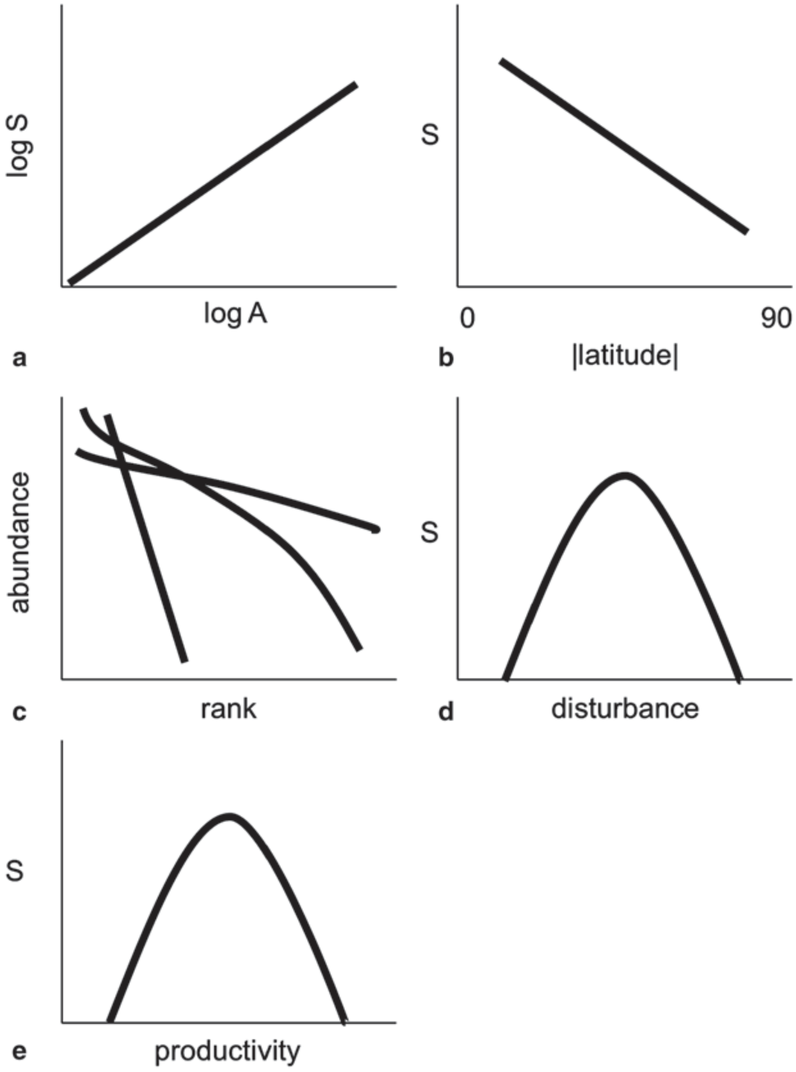

Fig. 7.1 **a** Species–area relationship. **b** Latitudinal gradient in biodiversity. **c** Dominance–diversity plots of relative abundance, where the *X*-axis represents species ranked in order of their abundance from common (near the origin) to rare (farther along the axis). **d** Intermediate disturbance hypothesis. **e** Diversity–productivity relationship

to a predisturbance state and urban development shifts the system to a new condition, effectively resetting the parameters as to what is "normal" and what is "disturbed." Even so, an IDH pattern is manifested in cities as an increase in diversity as one moves away from the centrally developed urban core, through a heterogeneous mosaic of medium-density development and habitat remnants towards the urban fringe that supports native and nonnative species alike, with a decline in diversity outside the city limits (Racey and Euler 1982; Blair 1999; Marzluff 2005).

Biodiversity tends to exhibit a unimodal relationship with primary productivity (a hump-shaped relationship akin to the IDH, where diversity is maximal at intermediate levels of productivity; Fig. 7.1e). However, there are numerous exceptions to this trend, and numerous potential mechanisms generating the various diversity–productivity relationships seen (Morin 2011). As a further complication, urbanization can decrease (Milesi et al. 2003; Xu et al. 2007) or increase net primary productivity (Buyantuyev and Wu 2009), with different trends possible at different spatial scales (Imhoff et al. 2004). A boost in urban diversity is typically attributed to the presence of nonnative, often invasive, species, and often at the expense of natives (Pyšek et al. 2004). Like the urban IDH, the urban diversity–productivity relationship may reflect changes in productivity along the urban–rural gradient from city center through residential suburbs to wildlands that are the result of both anthropogenic factors (such as decisions about yard composition) as well as non-anthropogenic ones (such as presence of impervious surfaces).

These aforementioned patterns result from abiotic filtering and interspecific interactions. Because communities are, by definition, multispecies assemblages, interactions among those species are key factors in determining community membership. Because urban development can generate novel assemblages of species with novel abiotic filters acting upon them, an examination of interspecific interactions is a key focus of urban community ecology.

7.3 Interspecific Interactions that Shape Communities

There are six types of biotic interactions that structure communities (including urban wildlife communities), defined on a pairwise basis of whether one of the participating species benefits from the interaction (+), is harmed (−), or neither (neutral, 0). These biotic interactions are competition (−/−), predation (+/−), parasitism (+/−), mutualism (+/+), commensalism (+/0), and amensalism (−/0).

Of these interactions, commensalism and amensalism are relatively rare and thus not strong forces shaping community structure. In contrast, competition has historically been considered to be the most important mechanism in structuring biotic communities, dictating diversity, abundance, and distribution patterns (Begon et al. 2006). Competition may involve direct aggression over limited resources (interference competition) or preemptive depletion of resources where the competitors may never come into direct contact (exploitative competition). Outcomes of competition include exclusion of one species from a community by another species, or coexistence if there is a change in the way resources are used such that resource overlap is reduced between the competitors. Selection for traits that promote resource partitioning can lead to character displacement, morphological changes that differentiate sympatric species so as to alleviate competition (Brown and Wilson 1956). Because cities bring together species that may not normally encounter each other (such as native and nonnative species) in settings with potentially elevated primary productivity, competition may be particularly fierce in urban ecosystems (Shochat et al. 2006).

Like competition, predator–prey relationships are affected by urban development. The loss of large carnivores in urban areas has led to an increase in mesopredators that were previously regulated by the presence of larger predators (Crooks and Soulé 1999; Crooks et al. 2010; Gehrt et al. 2010). These changes, combined with the occurrence of novel predator species in urban areas (e.g., domestic cats (*Felis catus*) and dogs (*Canis lupus familiaris*)), mean that urban food web structure is greatly modified (Faeth et al. 2005). Some implications of these changes will be explored in the next section.

Like predation, parasitism is a biotic interaction that involves benefit to one participant (the parasite) to the detriment of another (the host). The prevalence of parasites is often greater in urban than in rural populations, possibly due to crowding of hosts within urban areas, with increased host encounter rates leading to increased parasite transmission, relative to more dispersed hosts in wildlands (e.g., coyote heartworms; Gehrt and Riley 2010).

Mutualism includes interactions for mutual benefit in terms of resource acquisition, defense, or reproduction. The interactions of plants and their pollinators, soil mycorrhizae and plants, and the fungal–algal relationships that comprise lichens are all examples of mutualisms that are impacted by urbanization. For example, plant fitness has been shown to be compromised in some urban species because of reduced pollinator diversity (due to incidental mortality from pest control in urban areas, or to soil compaction for ground-nesting bees; Aizen and Feinsinger 2003; Liu and Koptur 2003). A similarly depauperate community of soil microorganisms is seen in urban areas due to changes in soil chemistry (Pouyat et al. 1994), particularly with respect to fungi (Newbound et al. 2010). In many European cities, a loss of mutualistic mycorrhizal partners eliminated many fungi species and altered overall plant composition (Arnolds 1991).

Based on these and other documented effects of urbanization on interspecific interactions, it is logical to question whether urban wildlife communities behave the same as do nonurban ones.

7.4 Do Urban Wildlife Communities Follow the Same Principles as "Traditional" Communities?

Cities were not widely recognized as ecosystems until the 1980s, well after the "golden age" of community ecology in the mid-1900s when foundational theories were developed and applied to address natural resource management needs. The paradigms of community ecology were constructed from study of areas with low evidence of human domination, as cities were seen as being antithetical to natural ecosystems. The history of community ecology thus begs the question, do urban communities behave as their country cousins do?

Just as cities change in structure and composition as they grow and age, so do biotic communities. In examining the effects of urbanization on community assembly and succession, most studies have used a customary "space for time" substitution by simultaneously examining areas representing different seral stages (e.g., recently disturbed areas, habitat remnants, and long-established districts). Our understanding of assembly rules in nonurban communities is muddled from long-standing debates about whether assembly is more deterministic than stochastic (Gotelli and McCabe 2002) and whether it follows a neutral model (Hubbell 2001; Ricklefs 2006). However, we do know that some communities are driven, at least in part, by chance, with membership determined via a "lottery" process of being lucky enough to occupy a membership vacancy (Sale 1978) or experience "carousel" dynamics where species richness, but not identity, may be predictable because species come and go (van der Maarel and Sykes 1993). Certainly all multispecies assemblages are generated via a filtration process whereby abiotic conditions, dispersal limitation, and interspecific interactions collectively dictate which species are present in a given area, and how predictably they occur. Urban communities likewise experience these filters but also must cope with four additional, anthropogenic filters (Williams et al. 2009): habitat transformation, habitat fragmentation, human alterations to the physical structure of the environment, and human decisions and preferences. Urban biodiversity is driven by factors unique to urban ecosystems (e.g., socioeconomic factors; Hope et al. 2003; Kinzig et al. 2005; Chap. 6) and thus uniquely reflects human decisions. For example, one unique urban pattern is a positive relationship between diversity and socioeconomic metrics associated with affluence, a "luxury effect" that is not seen in nonurban communities (Hope et al. 2003).

Urban communities often are simplified, comprised primarily of early successional species, ruderals, and nonnative species (Adams 1994), with concomitant changes in urban food web dynamics and simplified food web structure. Although some evidence has suggested that greater diversity is associated with greater system stability and ecosystem functioning (e.g., May 1972; Tilman and Downing 1994), some evidence suggests simpler systems may be more resistant to further invasion, resilient to disturbance, and more likely to persist (see reviews by McCann 2000; Hooper et al. 2005). The implications of these findings are that while urban communities may be less speciose and have fewer native species than nonurban communities, they may be highly durable.

Species richness for different taxa may be greater, lower, or unchanged in urban areas relative to rural sites, for a variety of reasons (Ricketts and Imhoff 2003; Pickett et al. 2008). Those species present in areas with reduced diversity are often early successional species or ruderals (Adams 1994; Stratford and Robinson 2005), adaptable generalists, and opportunistic species such as raccoons (Hadidian et al. 2010; Fig. 7.2). Although nonnative species may boost urban diversity, even native species may be attracted to urban areas because of an "oasis effect" (Bock et al. 2008), whereby natural fluctuations in resource availability are dampened, making food, water, and other important resources more constantly available (Shochat et al. 2004). The deliberate or inadvertent addition of resources (such as bird feeders, attics for bat roosts, or perches on towers and poles) may also boost urban diversity (Gehrt and Chelsvig 2003, 2004).

Species-abundance patterns in cities are usually much skewed, exhibiting dominance by species tolerant of human presence. These species are referred to as urban "exploiters" (using the terminology of McKinney 2002). Urban exploiters are

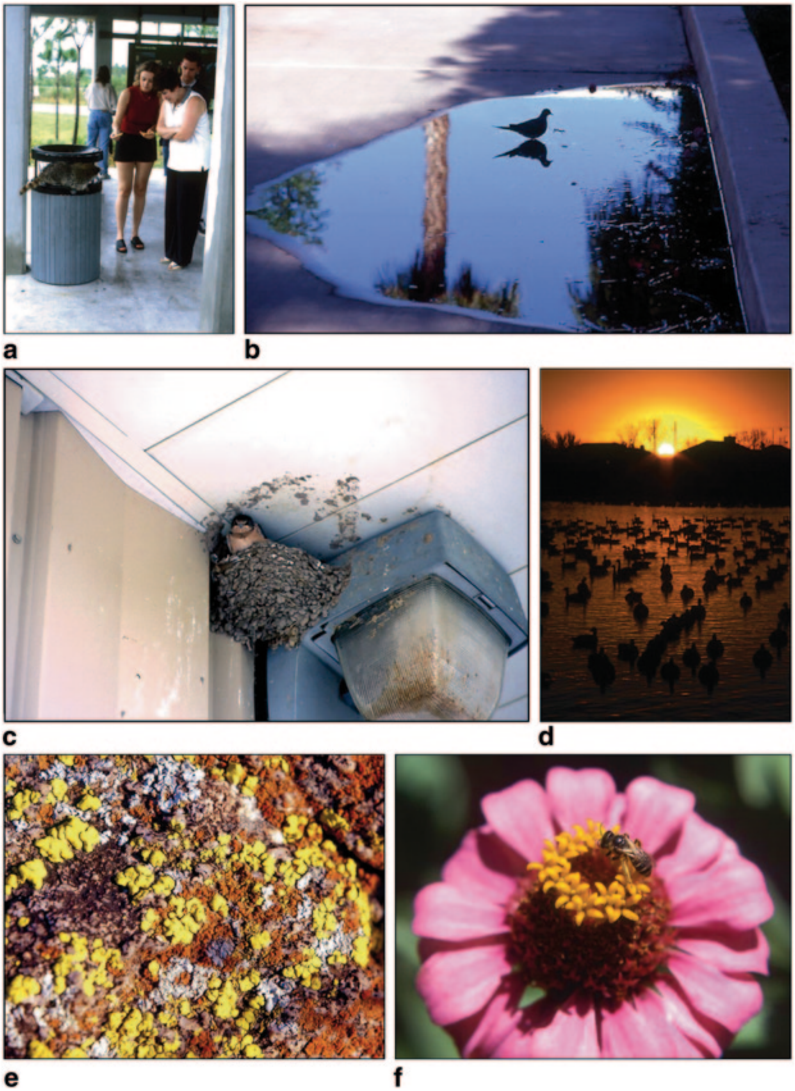

Fig. 7.2 Some examples of urban wildlife. **a** Raccoon raiding a trash can, with onlookers giving handouts (Florida, USA). **b** Mourning dove (Arizona, USA). **c** Barn swallow nest by a light fixture under a metal eave (Colorado, USA). **d** Waterfowl on an urban pond at sunset (Texas, USA). **e** Lichens (Arizona, USA). **b** Zinnia flower being pollinated by native bee (Arizona, USA). All photos by NEM

so well-adapted to built environments that they are seldom encountered outside of cities (e.g., pigeons, which formerly nested on cliffs and now make use of the "cliffs" that buildings provide). Not quite as closely associated with built environments, urban "adapters" (e.g., red foxes, gray squirrels, coyotes) also commonly occur in nonurban areas and have a generalist diet and high reproductive potential;

some such species even have increased their abundance or geographic distribution as a result of increased urban development (e.g., Mississippi kites, great-tailed grackles, Canada geese; Adams 1994). Urban "avoiders," in contrast, are seldom if ever encountered in urban areas; these include area-sensitive, interior species, species with large body sizes and species occupying high trophic levels.

Some synanthropic (urban-exploiter and urban-adapted) species may achieve abundances that are orders of magnitude greater than conspecifics in nonurban areas. For example, urban areas may support >300 raccoons per square kilometer compared to $20/\text{km}^2$ in rural sites (Lotze and Anderson 1979; Riley et al. 1998), with similar trends noted for gray squirrels (Parker and Nilon 2008) and many other species. A bottom–up view of urban resource exploitation in tandem with the loss of large carnivores in cities has been used to explain the highly skewed dominance–diversity relationships typically seen in urban areas (McCleery et al. 2008). Indeed, some predation-risk assessments of ground-feeding birds have concluded that they perceive less risk in urban than in rural environments, presumably due to a lack of urban predators (Shochat et al. 2004; cf. Valcarcel and Fernández-Juricic 2009). However, an increase in mesopredators due to large predator release (Crooks et al. 2010) as well as the introduction of domestic carnivores (cats, dogs) and presence of urban-adapter raptors (e.g., Cooper's hawks (*Accipiter cooperii*)) in urban areas would tend to support more top–down control than appears to occur (or at least be perceived by potential prey). It is possible that urbanization already filters out those prey species that are especially vulnerable to predation, or perhaps those species that are especially risk-averse. Just as the "ghost of competition past" (Connell 1980) results in assemblages of species that have experienced enough differentiation to alleviate competition and allow coexistence in the present day, so the "ghost of predation past" may result in predator-tolerant assemblages in urban environments (Shochat 2004). Because cities are relatively new, evolutionarily speaking, such effects may yet be developing in urban communities.

Whereas predation from large carnivores may not be as important a regulating factor in urban areas, other factors that are less common in other environments are important in urban ones, such as contaminant exposure or collisions with vehicles, towers, or windows. Similarly, urbanization may affect competition or parasitism by bringing species into contact with each other, including novel (nonnative) species, and at high densities in limited space. These factors may make urban areas into so-called ecological traps, areas that provide false cues of habitat quality. For example, Cooper's hawks may be attracted to prey availability and perching or nesting sites in urban areas but suffer greater nestling mortality due to infectious disease (Boal and Mannan 1999). But not all urban species appear to fall into such ecological traps (see e.g., Leston and Rodewald 2006).

Some of the differences between urban and nonurban communities may stem from inherent differences in their surroundings. For example, the urban heat island effect increases the length of the growing season (Baker et al. 2002; Zhang et al. 2004), which may increase the possibility of resource overlap and, thus, competition, in species that had formerly been temporally segregated. The presence of artificial lights may change the diet of nocturnal foragers like bats (Rydell 2006), roosting sites of birds (Gorenzel and Salmon 1995), or activity of herpetofauna (Perry et al. 2008) in cities. Finally, urban wildlife may develop a "junk food diet" that is quite literally based on garbage.

There are thus important differences in community structure between urban and nonurban ecosystems. Furthermore, because the factors that affect urban wildlife tend to be similar across cities around the world, urban communities tend to be similar in structure and composition, a pattern termed biotic homogenization (Blair 2001; Chace and Walsh 2006; McKinney 2006). These effects mean that natural resource managers must adapt traditional theories and methods to apply to urban communities, but that these adaptations will be broadly applicable across cities. Our global economy also is creating a global ecology; management of urban wildlife communities is, and will continue to be, an important and cosmopolitan activity.

Conclusion

Managing diversity is a key focus of wildlife biology, but it is an even more challenging endeavor when conducted in urban areas. Challenges arise both from complex and poorly understood inter-species relationships and from the difficulty of influencing human behaviors that drive many ecosystem processes. Effective planning and design for urban wildlife and for positive human–wildlife interactions must therefore include a social component (Chap. 6). Cross-disciplinary, team efforts among wildlife biologists, landscape architects, and land-use planners will become more common as urbanization occurs. Although these efforts will require unprecedented emphasis on communication (McIntyre 2011, Chap. 11), the commonalities across the globe of factors that structure urban communities may facilitate making action plans for urban biodiversity conservation. This chapter provides a basic understanding of community ecology needed to shape urban wildlife communities, while later chapters provide detailed examples of how to manipulate wildlife communities in urban areas and productively engage urban residents in those endeavors.

Community ecology is the science of biodiversity, interested in explaining how certain species can coexist whereas other combinations cannot, and why certain assemblages are stable whereas others are susceptible to invasion and alteration. Urbanization brings together species to create novel assemblages subject to novel regulation. As perhaps the main sign of the Anthropocene, urban ecosystems present wildlife scientists with many research opportunities that combine the familiar with the unique.

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Chapter 8
Wildlife Population Dynamics in Urban Landscapes

Amanda D. Rodewald and Stanley D. Gehrt

Keywords Age structure · Demography · Density · Disease · Genetic structure · Metapopulation · Mortality · Predation · Reproduction · Survival

To say that urban wildlife populations differ from those in other landscapes is, in some ways, to state the obvious, as many studies have well demonstrated how urbanization influences pattern of occurrence and relative abundance of wildlife (e.g., Blair 1996; Marzluff et al. 2001; Sinclair et al. 2005; Chace and Walsh 2006). Identifying the specific ways in which urban and nonurban wildlife populations differ and the drivers of those differences is less understood and requires a more careful examination. The response of a species to urbanization may be the consequence of life history, and behavioral and physiological attributes that promote avoidance, tolerance, or preference for urban systems. For example, urban avoiders, or species that respond negatively to development, tend to be habitat specialists, migratory, and/or sensitive to a wide range of human activities and disturbance (Croci et al. 2008). Species that respond positively to urban development (e.g., synanthropic species, urban exploiters) are often generalists, omnivorous, multi-brooded, and behaviorally flexible. Because these suites of species differ widely in population ecology even in the absence of urbanization, we forgo a direct comparison of these groups of species and, rather, examine how urbanization affects population structure and demography of species occupying both urban and nonurban landscapes (i.e., urban adapters).

As we synthesize the literature, we recognize the inherent difficulty of clearly defining a "population." What distinguishes a subpopulation from a population? For some species, this may be quite clear because of limited vagility and strong segregation among habitat types; for others, it may be unclear, particularly those with more generalist habitat requirements and extensive mobility across urban

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A. D. Rodewald (✉)
Cornell Lab of Ornithology and Department of Natural Resources, Cornell University,
Ithaca, NY 14850, USA
e-mail: arodewald@cornell.edu

S. D. Gehrt
School of Environment and Natural Resources, The Ohio State University,
Columbus, OH 43210, USA

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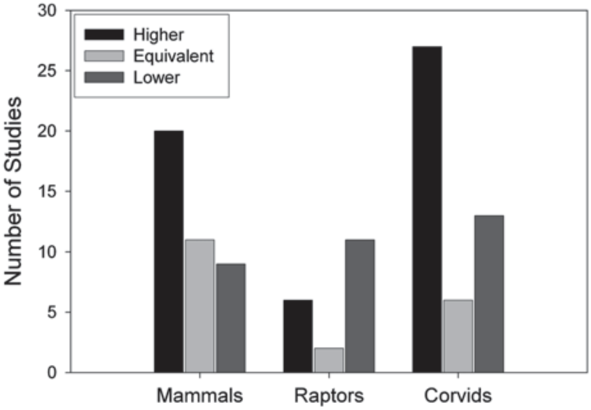


Fig. 8.1 Number of studies describing patterns of density in urban and nonurban wildlife populations. *Higher* represents higher densities in urban than rural areas, *equivalent* refers to no difference, and *lower* refers to species with lower densities in urban than rural areas. (Data from Fischer et al. 2012)

areas. Although we do not explicitly address population delimitation in our chapter, we alert the reader to the fact that defining a population becomes important when population demography and dynamics, such as vital rates, are the focus.

An important contextual backdrop to studies of urban wildlife is that most have focused on populations, or subpopulations, located within "greenspaces" within the larger urban matrix, or in some cases near the edge of the matrix. Many of the patterns we report are heavily influenced by the preponderance of studies within urban greenspaces, with relatively less attention given to populations living in the developed matrix. Processes we discuss may operate differently, and to different degrees, among populations occurring in the urban matrix proper. We begin this chapter with a review of patterns of population density that are often used to classify species and their relationships with urbanization. Next, we consider how urbanization affects the dynamics (survival and types of mortality, reproduction, and limiting factors) that influence variations in density. We also briefly review metapopulations and genetic structure in urban landscapes, and conclude with a discussion on the importance of understanding the linkages between urbanization and population dynamics for wildlife conservation and management.

8.1 Density

We begin our chapter with an examination of density because, as a noticeable characteristic of urban wildlife, it has been the most common, and oftentimes exclusive, focus of most studies of urban wildlife ecology. In addition, population density or abundance is often the primary indicator used to identify many species as urban exploiters or avoiders, and certainly density has important conservation and management implications.

Many wildlife species reach greater abundances and/or densities in urban than nonurban habitats (Fig. 8.1; Fischer et al. 2012; Møller et al. 2012). The range of

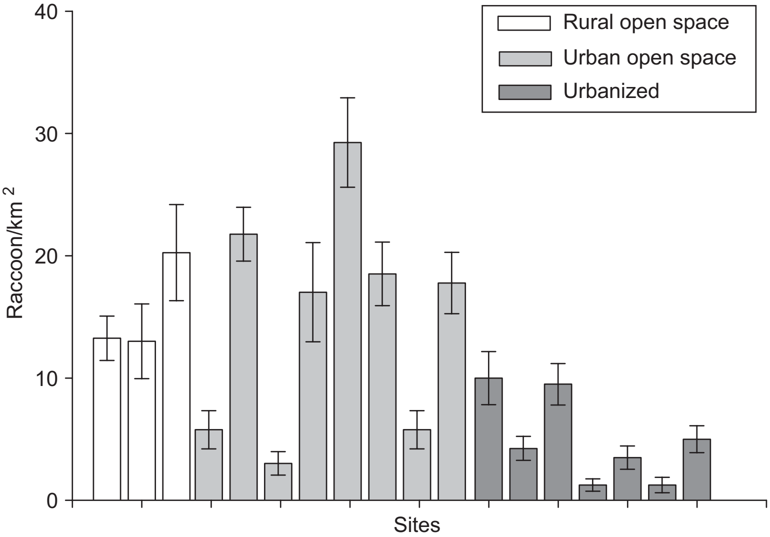


Fig. 8.2 Density estimates for populations of raccoons from 18 sites across the Chicago metropolitan area, based on mark-recapture data, 2005–2006. Populations were classified as occurring in rural natural fragments, urban natural fragments, and those within the urban matrix (urbanized), including industrial sites. (Adapted from Graser et al. 2012)

examples and magnitudes of the differences in density can be striking, with some species reaching 100 × greater densities in cities than rural areas (e.g., blackbird (*Turdus merula*) and magpie (*Pica pica*; Luniak et al. 1997). In general, the greatest shifts in density that occur among mammals in response to urbanization are those of medium-sized, omnivorous species, including European hedgehogs (*Erinaceus europeaus*) in France (Hubert et al. 2011), and brush-tail possums (*Trichosurus vulpecula*) in Australia (Stow et al. 2006). A classic urban adapter in the USA, the raccoon (*Procyon lotor*), can reach extremely high densities (> 100 raccoons/km^2) (Riley et al. 1998; Prange et al. 2003; Hadidian et al. 2010) that are about five times greater than typical nonurban densities (1–20 individuals/km^2, Gehrt 2003). However, most density estimates are from remnant habitats within cities and estimates for these same species outside preserved areas and within the larger urban matrix are less common. The few extensive mark-recapture studies conducted across urban landscapes, including developed areas, have produced highly variable density estimates. For example, raccoon densities ranged from 37 to 94 animals/km^2 across the Toronto, Ontario, area (Rosatte et al. 1992; Broadfoot et al. 2001), and 1 to 29 individuals/km^2 across the Chicago, Illinois, area (Graser et al. 2012). In both systems, densities were highest in forest fragments, and lowest in industrial or built areas, and illustrate the variability in abundance of a species that is often perceived to be ubiquitous across urban landscapes (Fig. 8.2).

On the other hand, decline in abundance of species less suited to anthropogenic disturbance is another hallmark of urban systems. For example, urban landscapes typically support comparatively few insectivorous and migratory birds relative to nonurban areas (Beissinger and Osborne 1982; Rodewald and Bakermans 2006). Snakes, too, tend to decline in number with urbanization (Patten and Bolger 2003), in part due to vehicular-related mortality and persecution (Akani et al. 2002). Large (>20 kg) mammalian carnivores may occur near the periphery of urban areas, but individuals using developed areas tend to be transitory if they occur at all (Crooks 2002; Iossa et al. 2010; Bateman and Fleming 2012). It is unclear for some species whether they increase or decrease with urbanization, as is the case for urban Eurasian badgers (*Meles meles*) that can occur at low (0–0.04/ha) or high (0.33/ha) densities compared to nonurban populations (Huck et al. 2008; Harris and Cresswell 1987; Harris et al. 2010).

Certain life history or ecological traits may facilitate changes in density. One emerging pattern is that generalist species with high fecundity, strong dispersal ability, omnivorous diets, and nonmigratory behaviors tend to have higher urban than rural densities (Bonier et al. 2007; Kark et al. 2007; Croci et al. 2008; Moller 2009). Yet these attributes fail to fully explain species-specific responses to urbanization, given that only 30–50 % of the variation in urban and rural densities was explained by life history and ecological traits (Evans et al. 2011).

At a more proximate level, patterns of densities across rural–urban systems may be driven by a variety of ecological factors associated with urbanization, including changes in food, vegetation, microclimate, and predators. Positive population responses are often attributed to food supplementation, which includes birdseed, trash, and even fruiting ornamental plants (Fedriani et al. 2001; Gehrt 2004; Prugh et al. 2009; Rodewald 2012). Not only can food supplementation affect density through survival and reproduction, but it can also be mediated through behavioral processes. For example, high density of American crows (*Corvus brachyrhynchos*) in cities was thought to be a behavioral response to rich anthropogenic resources that promoted numbers through reduced space needs of urban crows and immigration to the city by nonurban individuals (Marzluff et al. 2001). Likewise, the rich resources in cities make it possible for Virginia opossums (*Didelphis virginiana*) to maintain smaller home ranges, which can allow more packing of individuals into smaller spaces (Wright et al. 2012). The behavioral process of habitat selection also can drive densities when urban habitats contain more environmental cues used in habitat selection compared to nonurban areas (Leston and Rodewald 2006; Rodewald and Shustack 2008a, b). In cases where population attributes serve as cues (e.g., conspecific density) positive feedbacks can dramatically affect population dynamics and amplify increases or decreases in density (Schmidt et al. 2010).

Change in predation risk is another potential causal factor of urban-associated changes in density. For example, relaxed risk of predation is another frequently cited driver of high densities of prey and mid-trophic species (Faeth et al. 2005; Shochat et al. 2006). Because cities often lack the apex predators present in nonurban areas (Estes et al. 2011), mesopredator release has been suggested as a mechanism that allows mid-trophic predators to increase in number (Crooks and Soulè 1999; Ritchie and Johnson 2009; Prugh et al. 2009). Indeed, cities usually support greater numbers of some mesopredators (Sorace 2002; Prange and Gehrt 2004; Chace and

Walsh 2006; Rodewald et al. 2011; Fischer et al. 2012), and this may suggest that cities have increased risk of predation for small prey species when compared to nonurban areas (Stoate and Szczur 2006). Despite the conceptual appeal of this hypothesis (McKinney 2002), the mesopredator release hypothesis rests upon the assumption that populations of mid-sized predators are top–down regulated by predation—an assumption that has little or no empirical support for many of the native mesopredator species outside Canidae (Gehrt and Clark 2003; Prange and Gehrt 2007).

Ultimately, the mechanisms driving patterns in density are diverse and sometimes paradoxical. An excellent example is for coyotes in the Chicago area, for which high densities are the result of a combination of contrasting demographic and behavioral traits. The Chicago population exhibits much higher survival rates, and possibly higher reproductive rates, than outlying rural coyotes, so demographic processes are consistent with urban adapters. But these coyotes also exhibit behavioral characteristics more typical of urban avoiders, such as strong temporal and spatial avoidance of people and developed areas, enlarged home ranges in the urban matrix, and resistance to use of available anthropogenic foods or structures (Gehrt and Clark 2003; Prange and Gehrt 2007). Thus, to truly understand a relationship between a species and urbanization, one must go beyond measures of density and examine underlying demographic processes.

8.2 Demography

8.2.1 Age and Sex Structure

The sex and age composition of individuals in a population can provide important insights into the dynamics of the population. For example, populations with an age distribution biased toward young animals tend to have a high intrinsic growth rate, especially if survival is also high. For mammals, a population with a male-biased sex ratio may indicate the population serves as a sink, such as that reported for urban black bears (*Ursus americanus*) in the western USA, where large numbers of bears are attracted to urban food sources and killed at high rates (Beckmann and Berger 2003; Beckmann and Lackey 2008). In polygynous species, a high female-biased sex ratio in a population dominated by young age classes likely reflects a positive growth rate. For example, in an urban fox squirrel (*Sciurus niger*) population with high survival and reproductive rates, the juvenile-to-adult ratio of 0.44 was higher than for rural populations, with all factors combined suggesting that the urban squirrels comprised a source population (McCleery 2009).

In general, urban populations of mammalian carnivores tend to have similar age and sex structures to nonurban populations with relatively minor deviations (Prange et al. 2003; Gehrt and Riley 2010; Rosatte et al. 2010). Similarly, an urban population of Texas horned lizards (*Phrynosoma cornutum)* with apparent stationary growth had a nearly even (1M:1.05F) adult sex ratio, and the age ratio was 74% adult and 26% juvenile (Endriss et al. 2007), which was similar to a rural population (Montgomery and Mackessy 2003). However, in their examination of blackbirds across the western Palearctic, Evans et al. (2009) found that the proportion of first-year birds in urban populations was substantially lower than in rural ones—a pattern that they attributed to higher adult survival rather than reduced reproduction.

Perhaps more effort has been devoted to estimate local population structures of white-tailed deer (*Odocoileus virginianus*) than any other urban species, at least in the eastern half of North America. Such information is critical to develop management goals and to subsequently justify those goals to those segments of the public or policy makers concerned with deer management in their communities. Denicola et al. (2008) reported on the population demography of deer populations in four different urban areas that had been protected from harvest for > 10 years. Their results indicated that sex and age structure of nonhunted deer populations may be fairly predictable, such that a typical nonhunted, suburban population is slightly biased toward females (60F:40M), with an age structure of 40% yearlings-to-adults for females and 20% yearlings-to-adults for males, and an overall age structure of the population is made up of 40% fawns to yearling-adult age classes.

8.2.2 Survival

When densities differ between urban and nonurban populations, the pattern is often, though not always, a product of shifts in survival or reproductive rates (Fig. 8.3). Perhaps owing to the difficulty in studying small populations, few studies have documented lower survival rates for species that are negatively associated with urbanization (but see Price et al. 2011). In contrast, several studies provide evidence that elevated survival rates in cities (e.g., Gosselink et al. 2007; McCleery et al. 2008; Lehrer et al. 2012) promote high densities of species, as with raccoons (Prange et al. 2003), fox squirrels (McCleery et al. 2008), eastern long-necked turtles (*Chelodina longicollis;* Rees et al. 2009), and many birds (e.g., Canada geese (*Branta canadensis; Balkcom* 2010); northern mockingbirds (*Mimus polyglottos;* Stracey and Robinson 2012), peregrine falcons (*Falco peregrines anatum;* Kauffman et al. 2003), and great tits (*Parus major;* Horak and Lebreton 1998)). On the other hand, a review of carnivores showed equivocal comparisons of survival rates between urban and rural populations, with some species exhibiting greater survival with urbanization (i.e., kit foxes (*Vulpes macrotis*), raccoons, coyotes, stone marten (*Martes foina*)) and others with no change or negative trends (i.e., striped skunks, red foxes, bobcats, mountain lions (*Puma concolor*; Iossa et al. 2010)). Nor are the high densities of urban northern cardinals (*Cardinalis cardinalis*) explained by demography alone, as survival rates of adults (Rodewald and Shustack 2008a, b) and juveniles (Ausprey and Rodewald 2011) were similar for urban and rural individuals (Fig. 8.4).

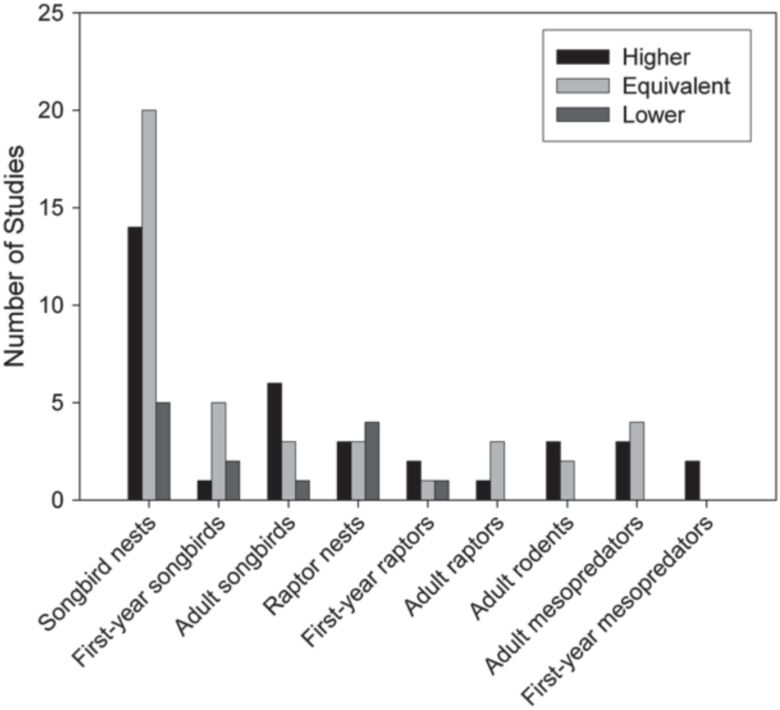
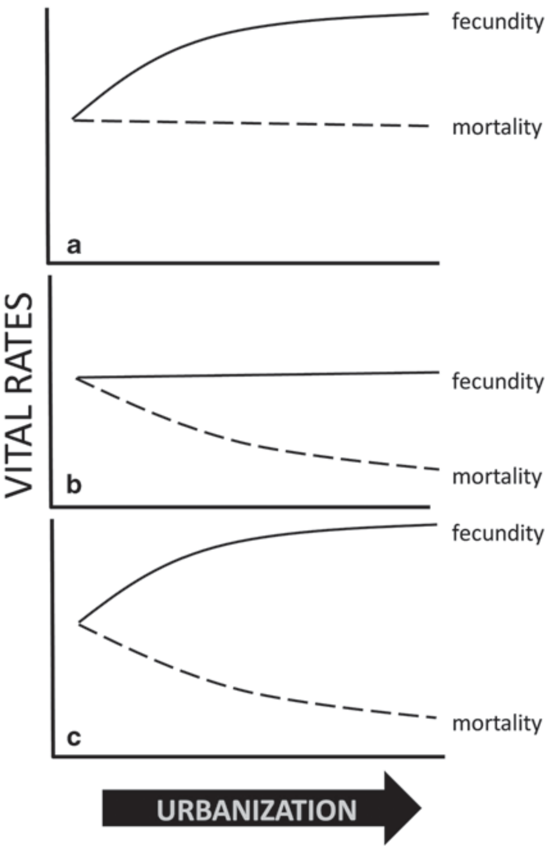


Fig. 8.3 Numbers of studies indicating higher, lower, or equivalent rates of survival in urban compared to rural habitats. (Adapted from Fischer et al. 2012)

Complicating the matter is the fact that there may be high spatiotemporal variation in survival rates across different life stages. For example, adult and nestling survival for Florida burrowing owls (*Athene cunicularia floridana*) declined along a development gradient, but juvenile survival improved (Millsap and Bear 2000; Millsap 2002). Survival rates of songbird nests and juveniles tended to be equivalent, but adult survivorship generally greater, in urban than rural habitats (Fig. 8.3; Fischer et al. 2012). Similarly, in urban areas, survival of white-tailed deer fawns and black bear cubs were lower than adults (Beckmann and Lackey 2008; Piccolo et al. 2010), and juvenile raccoons may have a greater susceptibility to vehicle collisions than adults (Hadidian et al. 2010). Heterogeneity in urban-associated ecological changes can produce spatial variation in survival rates even within cities. Gray catbirds (*Dumatella carolinensis*) illustrate how heterogeneity in urban-associated ecological changes can drive spatial patterning in juvenile mortality, which was most related to, and depressed by, local abundance of domestic cats (*Felis catus*; Balogh et al. 2011). There also can be striking differences in survival rates between the developed matrix and remnant habitat patches within cities (Whittaker and Marzluff 2009; but see Rosatte 2000; Gross et al. 2012; Gehrt unpublished data).

Fig. 8.4 Demographic pathways to synanthropy. Scenarios focusing on shifts within reproductive or survival rates along an urban gradient. Each scenario begins with a stable population with equal mortality and fecundity rates in a nonurban landscape. A potential increase in density or emigration increases with urbanization when fecundity increases (**a**), overall mortality decreases (**b**), or both (**c**). We would predict species that exhibit the greatest degree of synanthropy, such as artificially-high densities, to occur under **c**



8.2.3 *Mortality Causes*

Sources of mortality can vary widely between urban and nonurban populations, and can also vary dramatically within the city between exclusively terrestrial and flying species, the latter of which are more vulnerable to collisions with buildings and windows. The death toll of migrating birds attracted to tall lighted buildings can be staggering, especially in foggy conditions (Longcore and Rich 2004). Even for local breeding populations, collisions can have important population-level consequences. For example, the greatest cause of mortality of Cooper's hawks (*Accipiter cooperii*) in Tuscon, Arizona was collisions (70 %), primarily with windows (Boal and Mannan 1999).

8.2.3.1 Roads

Arguably the greatest challenge for exclusively terrestrial species to overcome in exploiting the urban landscape is roads and the associated vehicles. The implications that roads have for urban wildlife are addressed in more detail in another chapter (Chap. 15), but we briefly summarize its relevance to population demography here. The importance of vehicles as a mortality source across a range of terrestrial taxa is shown in Table 8.1. In some cases, vehicle mortality is so high that it represents the greatest limitation to population growth, and may exceed reproduction, such as for black bears (Beckmann and Lackey 2008) and some reptiles (Cureton and Deaton 2012). Some turtle species are highly susceptible to road mortality because they are not able to adjust or avoid road traffic, and their relatively low fecundity does not allow for replacement of individuals killed by cars. Turtle populations affected by road mortality are those that are male biased and occur at low densities (Cureton and Deaton 2012).

Some urban adapted species appear to make behavioral adjustments to roads and traffic. Although vehicle-related mortality is common for white-tailed deer, raccoons, kit foxes and coyotes (Table 8.1), each of these species appear to reduce the risk of collision by avoiding roads or avoiding traffic when they do cross (Etter et al. 2002; Cypher 2010; Gehrt unpublished data). Notably, raccoons and coyotes that live exclusively in the urban matrix, and consequently cross many roads regularly, have the same risk of collision as individuals living in the urban parks that are exposed to fewer roads and less traffic (Gross et al. 2011; Gehrt unpublished data). However, in contrast to deer and coyotes, juvenile raccoons may be susceptible to vehicle mortality and this may be an important density-independent limitation to their population growth (Hadidian et al. 2010).

8.2.3.2 Predation

The combined effects of urban habitat fragmentation and elevated native and nonnative predator abundance can produce high levels of predation on terrestrial and flying species alike. Predation levels from mammalian mesopredators can threaten herpetofauna and shift community structure in suburban and urban aquatic systems (Mitro 2003; Eskew et al. 2010). Increased predation levels on avian populations are well-documented in some, but not all urban systems (Chamberlain et al. 2009; Fischer et al. 2012). Large numbers of nonnative species, such as domestic cats, prey on small mammal and avian species (Beckerman et al. 2007; van Heezik et al. 2010; Loss et al. 2013). Predation rates for some species may decline with urbanization, especially for mammals such as rodents (Table 8.1). For example, woodchucks (*Marmota monax;* Lehrer et al. 2012) and fox squirrels (McCleery et al. 2008) are more likely to die from predation in rural areas, but mortality risk shifts from predators to vehicle collisions with urbanization. Likewise, coyote killing of foxes, due to intraguild competition, decreases with urbanization in those cities where coyotes are not as abundant as adjacent rural areas (Gosselink et al. 2007; Cypher 2010). The noise of cities also may indirectly reduce predation in cases where predators avoid the loud environments (Francis et al. 2009).

Nest predation has probably been better studied than any other type of predation in urban areas. Although cities generally have greater numbers of generalist and opportunistic predators that prey upon bird nests (Sorace 2002; Prange and Geh-

Table 8.1 Cause-specific mortality for terrestrial vertebrates derived from radiotelemetry. Number of mortalities (*n*) and the percentage of mortalities by three types of mortality. We have highlighted three types of mortality, therefore total percentages for each species will not be equal to 100 %. The location of the study site is noted as occurring within a habitat fragment (urban fragment), within the urban development or matrix (urban matrix), or a combination of fragments and matrix (mixed)

Species	*n*	Landscape type	Vehicles (%)	Predation (%)	Disease (%)	Source
Horned lizard	9	Urban fragment	11	44	0	Endriss et al. (2007)
Fox squirrel	26	Urban fragment	57	4	0	McCleery et al. (2008)
Woodchuck[a]	26	Mixed	19	27	0	Lehrer et al. (2012)
Red fox	173	Mixed	36	9	39	Gosselink et al. (2007)
Red fox (premange)	80	Mixed	62	0	17	Soulsbury et al. (2010)
Red fox (postmange)	67	Mixed	32	0	61	Soulsbury et al. (2010)
Kit fox	56	Mixed	45	30	0	Cypher (2010)
Raccoon	18	Urban fragment	17	0	50	Riley et al. (1998)
Raccoon	13	Urban fragment	23	0	77	Prange et al. (2003)
Raccoon	18	Suburban fragment	56	0	39	Prange et al. (2003)
Raccoon	16	Urban matrix	56	0	19	Gross et al. (2012)
Striped skunk	23	Urban fragment	17	0	70	Gehrt (2005)
Bobcat	49	Mixed	69	0	22	Riley et al. (2010)
Coyote	68	Mixed	62	0	10	Gehrt et al. (2011)
Black bear	156	Mixed	57	0	0	Beckmann and Lackey (2008)
White-tailed deer	40	Suburban fragment	72	0	0	Etter et al. (2002)

[a] Study occurred along an urban–rural gradient; mortality was primarily vehicles, and less due to predation, for urban animals (Lehrer et al. 2012)

rt 2004; Rodewald et al. 2011; Fischer et al. 2012), there is paradox in that links between urbanization and nest predation are surprisingly weak with mixed empirical support (Chamberlain et al. 2009; Fischer et al. 2012). Rural nest survival has been similar (Reidy et al. 2009; Burhans and Thompson 2006; Rodewald et al. 2013), greater (Newhouse et al. 2008; Ryder et al. 2010; Stracey and Robinson 2012), and lower (Phillips et al. 2005; Vigallon and Marzluff 2005; Bakermans and Rodewald 2006) than urban nest survival. Part of the variation may result from different scales of study, as research examining remnant patches of habitat in urban and rural landscapes usually finds similar rates of nest predation, whereas research comparing developed and undeveloped areas within the urban matrix finds higher nest survival within the developed areas (Fischer et al. 2012). Ongoing research comparing nest survival in suburban yards and adjacent forest parks shows that early-season nest survival is greater within the developed matrix than the remnant forest patches even within the city alone (J. Malpass and A. Rodewald, unpublished data). Temporal patterns of nest depredation also can vary with urbanization. Suburban Florida scrub-jays (*Aphelocoma coerulescens*) experienced lower rates of depredation during egg stage but higher depredation during nestling stage than nonurban populations, though overall rates of success were similar (Bowman and Woolfenden 2001).

The "predator paradox", where high predator numbers in cities are not matched with correspondingly high rates of nest predation, is supported both empirically in demographic studies (Rodewald et al. 2011; Stracey 2011) as well as in literature reviews (Fischer et al. 2012). A 10-year study of nearly 5000 nests of five songbird species breeding in forests along a rural-to-urban gradient in Ohio provides an excellent illustration. The nest predator community of the study system was diverse with 21 video-documented predator species, most of which were generalist species (Rodewald and Kearns 2011). Despite greater numbers of nearly all documented predators within urban compared to rural landscapes, there were no consistent relationships between avian nest survival and urbanization for any of the focal bird species (Rodewald et al. 2013). Moreover, although increasing predator detections were associated with nest survival in rural landscapes, predator-prey relationships were decoupled in urban landscapes such that predator activity failed to predict nest survival rate. Similar patterns have been detected in European cities, where superabundant hooded crows (*Corvus cornix*) (Weidinger 2009) and magpies (Chiron and Julliard 2007) were less important nest predators than in nonurban landscapes. This apparent disconnect between rates of nest predation and predator activity in urban landscapes may arise because many urban predators are heavily subsidized by anthropogenic food sources (Gehrt 2004; Prange et al. 2004; Marzluff and Neatherlin 2006; Withey and Marzluff 2009; Rodewald et al. 2011) and therefore may depredate fewer nests than less subsidized rural predators.

There is growing evidence that the species most responsible for nest depredation differs between urban and nonurban areas. Nests of understory birds, including northern cardinal, Acadian flycatcher (*Empidonax virescens*), gray catbird, and wood thrush (*Hylocichla mustelina*), were three times more likely to be depredated by mesopredators in cities compared to rural areas (35 % vs. 13 % of nests; Rodewald and Kearns 2011). On the other hand, rural nests were most likely to be depredated by small passerine birds (e.g., brown-headed cowbird (*Molothrus ater*) and common grackle (*Quiscalus quiscula*) and raptors. Reidy et al. (2008) also documented that the dominant avian nest predator of golden-cheeked warbler (*Dendroica chrysoparia*) shifted from the western scrub-jay (*Aphelocoma californica*) in urban areas to the American crow in rural landscapes. Domestic cats were the most important predators of northern mockingbird nests (Stracey 2011) and juvenile gray catbirds within cities (Balogh et al. 2011). However, the impact of cats may not be similarly distributed across cities. Ongoing work video-documenting nest predation in forest parks vs. adjacent backyards shows that domestic cats are more likely to depredate nests in yards than nests in adjacent forests (J. Malpass and A. Rodewald, unpublished data).

8.2.3.3 Disease

For most mammals, disease influences population dynamics in all systems to varying degrees, but in urban systems it is most apparent for mammalian mesopredators (Table 8.1, see Chap. 10), and it usually takes the form of transmissible diseases because of artificially-high host densities and consequent interactions between individuals. For example, rabies is an important disease that can dramatically reduce host density in urban areas and other notable epizootics in urban areas have been reported for raccoons, striped skunks, and red foxes (Riley et al. 1998; Rosatte 2000). Much like rabies, epizootics of sarcoptic mange (*Sarcoptes scabei*) impact high-density canid populations, and these can be dramatic in urban areas where populations are subsidized with anthropogenic foods and buffered from predation (Gosselink et al. 2007; Soulsbury et al. 2010).

Canine distemper is a morbillivirus commonly found in urban mesopredator populations, especially raccoons and skunks (Gehrt 2004). In fact, canine distemper virus is probably enzootic in most urban North American raccoon populations as a result of high host densities (Roscoe 1993; Gehrt 2003), and periodic outbreaks of new strains may occur that impact domestic animals as well as native hosts (Cleaveland et al. 2000; Hadidian et al. 2010).

Although transmissible diseases may have less impact on urban avian populations than predation or accidents, novel pathogens introduced to urban systems can severely limit populations of some species, as in the case of West Nile Virus and American crow populations (*Corvus brachyrhynchos*; LaDeau et al. 2007). An alarming 72 % of the crow population in Stillwater, Oklahoma, was lost to this disease in 1 year (Caffrey et al. 2005). Moreover, avian exposure to West Nile Virus can be greater within urban than nonurban areas, as shown in the Chicago region (Hamer et al. 2012). A recent literature review showed that urbanization could be positively or negatively associated with the diversity and prevalence of bird parasites (Delgado and French 2012). Comparing urban and rural blackbird populations across Europe, Evans et al. (2009) found that prevalence of both ticks and avian malaria was lower in cities.

8.2.3.4 Other Mortalities

Urban wildlife die from many causes in addition to those mentioned above, including electrocution, drowning, poisoning, nuisance removal, and entombment, but we have highlighted those that have been reported to have population-level effects. However, one cause of mortality that is notable because of its *absence* from urban systems is harvest or the hunting and trapping of game species. In North America, most of the mammals common to urban systems, and many that are considered overabundant, are game animals with legal harvest outside of cities. The annual harvest of some of these species is substantial; for example, during a single year (the 2010–2011 harvest season) 110,415 opossums, 216,663 coyotes, 74,223 striped skunks, and an impressive 801,335 raccoons were harvested across 13 states in the Midwestern USA (source: Association of Fish and Wildlife Agencies). Indeed, across the USA approximately 1 million raccoons are harvested from primarily rural areas each year. For these species it is harvest, rather than predation from apex predators, that typically represents the primary form of mortality in rural areas, in contrast to the frequent assumption by others claiming mesopredator release as a primary mechanism (McKinney 2002).

Similarly, for white-tailed deer harvest is the primary cause of death in rural areas (Nixon et al. 2001). For example, hunting-related mortality of white-tailed deer in rural Illinois makes up 60 % of all causes of mortality for does, and 78 % for bucks, of all ages, compared to 18 and 13 % (does and bucks, respectively) vehicle-related mortality. Consequently, annual survival across subadult-adult age classes ranged 56–92 % for does and 35–76 % for bucks (Nixon et al. 2001), which was much lower than annual survival rates for deer in suburban Chicago where auto collision was the primary cause of mortality (Etter et al. 2002). The same situation likely occurs for waterfowl, particularly giant Canada geese that become nonmigratory in urban systems. Thus, it is not surprising that generalist species already well-equipped for urban landscapes exhibit accelerated population growth and attain relatively high densities when the primary mortality agents are removed.

8.2.3.5 General Mortality Patterns

The highly generalized patterns that emerge from reviews of the mammalian and avian literature to date (Chamberlain et al. 2009; Fischer et al. 2012, this chapter) reveal that for either taxa, anthropogenic-related mortality increases quickly with urbanization as would be expected (Fig. 8.5), although the specific causes may differ between the groups (i.e., vehicles for mammals, buildings and other structures for some birds). However, mortality rates associated with predation and disease tend to change more strongly with urbanization for mammals than for birds (Fig. 8.5). For mammals, there is a “strong” negative curve as predation or more likely harvest declines or disappears with urbanization consistently across species, whereas changes in predation rate for birds are mixed across species and systems, with predation often remaining an important form of mortality even if it trends slightly downward with urbanization. Disease often exceeds predation as a mortality factor for urban mammals, whereas disease does not commonly exceed predation for urban birds. These are generalizations, and the relative risk of predation or

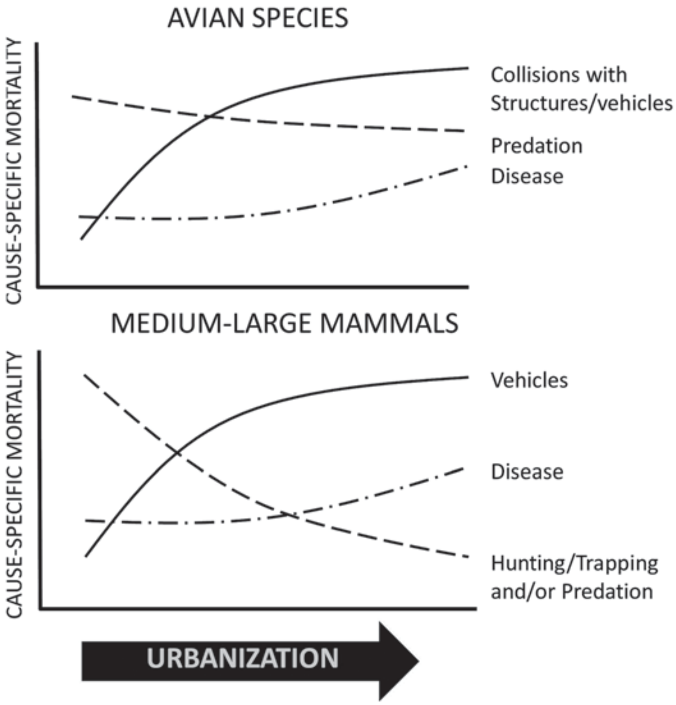


Fig. 8.5 Generalized trends for causes of mortality among urban species, as one moves along a rural-to-urban gradient. The curves are merely conceptual based on the literature and do not represent a formal meta-analysis. Curves exhibiting extreme changes in mortality rates (such as vehicles) represent the "strength" (or consistency) of the pattern among studies, whereas a slight curve (such as disease) reflects mixed results and a more complex relationship with urbanization

disease may change with life stage, such as higher predation for nestlings/fledglings than for adults, whereas risk from disease may be greater for adults.

8.2.4 Reproduction

8.2.4.1 Phenology

Urban populations often initiate breeding earlier in the season than their rural counterparts. A review of avian reproductive phenology by Chamberlain et al. (2009) showed that urban egg-laying was earlier in 16 of 19 studies. Although breeding early is thought to improve reproductive success (Perrins 1970; Norris 1993; Verhulst et al. 1995), some urban breeders may advance reproductive timing by 7–10 days with no detectable benefit in terms of reproductive output (Shustack and Rodewald 2011). Although the federally threatened Florida scrub-jay initiated breeding earlier in suburban than rural populations due to the protein-rich anthropogenic subsidies in suburban areas (Schoech and Bowman 2003; Fleischer et al. 2003), advanced laying dates resulted in greater hatching failures due to increased exposure of eggs to low ambient temperatures (Aldredge et al. 2012). While most studies have demonstrated advanced phenology of residents and short-distance migratory birds in cities (Eden 1985; Antonov and Atanasova 2003; Schoech and Bowman 2003), urban birds also may initiate breeding later. At higher levels of urbanization, Acadian flycatchers, a Neotropical migratory bird, arrived and initiated clutches later and finished nesting earlier (due fewer breeding attempts) than in forests within more rural landscapes (Rodewald and Shustack 2008b; Shustack and Rodewald 2010).

The causes of phenological shifts are unknown for most species but include a wide variety of social and ecological factors. In the aforementioned study of delayed reproduction for urban Acadian flycatchers, evidence suggested that urban forests were less desirable and selected later by smaller individuals, which initiated first clutches later in the nesting season than larger birds (Shustack and Rodewald 2010). For European blackbirds, a captive-rearing experiment indicated that earlier breeding in cities was primarily due to plasticity in hormone secretion and gonadal development (Partecke et al. 2004) likely related to social interactions and environmental conditions (Partecke et al. 2005). Artificial night lighting was implicated as the driver of advanced phenology of five common forest-breeding songbirds, with blue tits laying eggs 1.5 days earlier when territories included street lights than without (Kempenaers et al. 2010). Environmental conditions related to climate (e.g., temperature accumulation, leaf phenology) explained the timing of breeding of northern cardinals (Shustack 2008) and great tits and blue tits (*Parus caeruleus*; Nilsson and Kallander 2006).

8.2.4.2 Reproductive Output

Even if rates of nest predation are no higher in urban than nonurban habitats, reproductive output can be lower within cities due to other factors. In the central Ohio system, urban Acadian flycatchers produce fewer young annually despite similar rates of nest predation in urban and rural forests due to a combination of flycatcher behavior (i.e., fewer nest attempts) and brood parasitism (Rodewald and Shustack 2008b; Shustack and Rodewald 2010). Despite increased breeding densities within cities, reproductive performance within urban areas was lower than those surrounded by rural land for great tits (Hedblom and Soderstrom 2012), American crow (Marzluff et al. 2001), and house sparrows (*Passer domesticus*; Schroeder et al. 2012; Seress et al. 2012). Differences in productivity may reflect poorer food quality or nutrient restriction in urban compared to nonurban habitats (Solonen 2001; Antonov and Atanasova 2003; Heiss et al. 2009; Ibanez-Alamo and Soler 2010).

For some species, reproductive performance is greater within than outside of metropolitan areas for other species. In their review of the avian literature, Chamberlain et al. (2009) reported that annual productivity was greater in urban than nonurban habitats for seven of eight urban adapting species though nestling weight was lower in urban than rural areas for nine of ten urban adapters. Brood size of northern goshawks (*Accipeter gentilis*; Solonen and Ursin 2008) as well as number of young fledged by northern mockingbirds (Stracey and Robinson 2012) and house wrens (*Troglodytes aedon*; Newhouse et al. 2008) was greater in urban than rural sites. Urban adapting mammalian species that are able to exploit anthropogenic resources also tend to have elevated reproductive rates compared to rural populations. Reproduction was relatively higher for urban black bears (Beckmann and Lackey 2008), white-tailed deer (Etter et al. 2002), raccoons (Prange et al. 2003), kit foxes (Cypher 2010), San Clemente Island foxes (*Urocyon littoralis clementae*, Gould and Andelt 2011), and fox squirrels (McCleery 2009) than for rural populations. Apparently the artificially-abundant food allows females to continue to reproduce without density-dependent effects manifesting at population sizes similar to those found in nonurban areas. For example, it is well-established that canid reproductive success is closely tied to food abundance, and small-to-medium sized canids readily use anthropogenic foods in urban areas (Iossa et al. 2010). Higher fecundity for urban kit foxes than for nonurban foxes is attributed to the constant, predictable supply of anthropogenic foods in urban areas compared to the highly variable natural prey supply in natural or rural systems (Cypher 2010). Although reproductive output is often treated as a useful indicator of habitat quality, an important caveat is that for species occurring at densities near carrying capacity, as may be the case for urban-adapted species, density-dependent regulatory mechanisms may result in equivalent reproductive rates in urban and rural habitats (Rodewald and Shustack 2008a).

A New Classification System for Urban Wildlife Seth P. D. Riley and Stanley D. Gehrt

Urban ecologists frequently classify wildlife based on their relationship to urbanization. Throughout this book, you will see the use of a three-category system of urban exploiters, urban adapters (originally "suburban adaptable" species), and urban avoiders, developed by Blair (1996, 2001) for urban birds and later adapted for other taxa (McKinney 2002). This system has three problems: (1) coarse partitioning, (2) exclusive partitioning, and (3) "urban adapter" has erroneous evolutionary connotations. Accordingly, we propose a new way of categorizing wildlife to address these problems. This system replaces the original three categories with four to facilitate greater precision in understanding how different wildlife populations interact with urban processes. In contrast to the commonly used classifications, our categories are meant to be fluid in relation to individuals, species, populations, and time; the same species may belong to different categories in different situations, different individuals within the same population may belong in different categories, and specific individuals may even belong in different categories at different times in their lives.

Our proposed classification of species in urban areas is:

Urban dependents**:** Wildlife that are dependent on humans for food and shelter, even though humans do not provide it intentionally (as they do for livestock and pets). These animals are small and cryptic enough (rats and mice) or mobile enough (pigeons and house sparrows) to avoid humans even in urban cores.

Examples: house mouse (*Mus musculus*), Norway rat (*Rattus norvegicus*), rock pigeon (*Columba livia*), house sparrow (*Passer domesticus*).

Location: Typically found at greatest densities in or near the urban core, quickly decline in occurrence in the suburbs, and generally are rare in natural landscapes.

Urban exploiters**:** Wildlife able to exploit anthropogenic resources available in urban areas, but not dependent on these resources. These animals typically have generalized ecological niches with flexible behavior that allows exploitation of food and shelter that humans provide.

Examples: raccoon (*Procyon lotor*), coyote (*Canis latrans*), red fox (*Vulpes vulpes*), stone marten (*Martes foina*), eastern cottontail (*Sylvilagus floridanus*), gray squirrel (*Sciurus carolinensis*), white-footed mouse (*Peromyscus leucopus*), house finch (*Haemorhous mexicanu*s), peregrine falcon (*Falco peregrinus*), northern cardinal (*Cardinalis cardinalis*), house gecko (*Hemidactylusfrenatus*).

Location: Found to varying degrees throughout the urban landscape, but likely reach greatest densities in areas with green fragments interspersed within the urban matrix, or in low to medium density residential areas. Can reach greater densities in urban landscapes than in nonurban ones.

Urban tolerant**:** Wildlife that may use some anthropogenic resources and may reside in parts of the urban landscapes but do not generally exploit urban areas to reach greater densities. In some cases, density may be misleading in that urban populations may be sinks, with relatively high mortality or low reproduction.

Examples: bobcat (*Lynx rufus*), black bear (*Ursus americanus*), leopard (*Panthera pardus*) in India, some bat species, white-tailed (*Odocoileus virginianus*) and mule (*Odocoileus hemionus*) deer (but may sometimes be exploiters), some songbirds (e.g., red-eyed vireo (*Vireo olivaceus*), generalist or cryptic reptile and amphibian species such as American toad (*Anaxyrus americanus*), brown snake (*Storeria decayi*), green anole (*Anolis carolinensis*), legless lizards (*Anniella pulchra*), alligator lizards (*Elgaria* spp.), some small plethodontid salamanders (*Batrachoseps attenuatus* and *B. nigrventris* in California, *Plethodon cinereus* in the eastern USA).

Location: They generally occur in low density residential areas with vegetated yards, or medium density residential areas in close proximity to patches of natural landscape. Their occurrence declines quickly toward the urban core.

Urban avoiders**:** Wildlife that have narrow ecological niches or other traits that conflict with urbanization. These animals may occur at the margins of urban areas, but rarely occur in suburban or urban zones, unless large natural fragments occur near edges. If they are in urban landscapes, it is generally temporary or a transient individual (although over time, an individual may become an "urban tolerant.")

Examples: mountain lion (*Puma concolor*), gray wolf (*Canis lupus*), grizzly bear (*Ursus arctos horribilis*), many native small mammals, habitat-specific birds associated with recent burn areas (e.g., Bachman's sparrow, *Peucaea aestivalis*), early-succession areas (e.g., prairie warbler (*Setophaga discolor*)), or longleaf pine forest (e.g., red-cockaded woodpeckers, *Picoides borealis*).

Location: If they are found in urban or suburban areas, it's generally near open space.

Finally, there are some species that are likely never to be found in urban landscapes or even close to urban areas, even transient individuals, because they are very specific to certain natural habitats and highly sensitive to anthropogenic processes including habitat fragmentation, noise, and altered ecological systems. This group, which we would call "urban impossibles," might include species such as tailed frogs (*Ascaphus truei*) that require cold streams in old growth northwest forest, spotted owl (*Strix occidentalis*), mountain gorilla (*Gorilla beringei beringei*) in Africa, or snow leopard (*Panthera uncia*) in Asia. However, over time, even these species may come increasingly into contact with urban areas (Fig. 8.6).

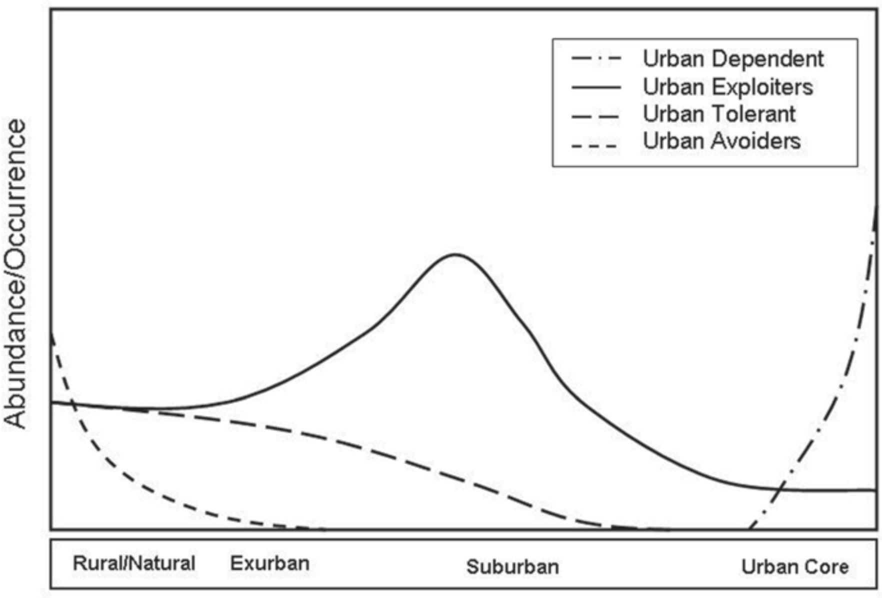


Fig. 8.6 Conceptual model illustrating the ways that wildlife species respond to urbanization, as indicated by abundance across the urban gradient

8.3 Limiting Factors

8.3.1 Density-Independent Factors

It has been long understood that populations are regularly affected by density-independent and density-dependent limiting factors. In nonurban systems, density-independent factors typically involve weather or other environmental conditions. Within urban systems, many of these traditional factors are ameliorated to various degrees. For example, drought in nonurban systems is buffered in the urban landscape with artificial sources of free-standing water, and climate-induced swings in natural food availability are buffered by anthropogenic foods. This overall pattern of buffering from extreme environmental conditions may explain why few urban wildlife studies have identified density-independent factors as important for limiting populations. One notable exception would be the heat island effect in large urban centers reducing the limiting effect of winter weather at northern latitudes for some species (e.g., opossums, Kanda et al. 2009). Other possible density-independent factors may include roads and toxins, such as rodenticides (Chaps. 10, 15).

8.3.2 Density-Dependent Factors

Many urban adapters occur at relatively high population densities, so it may seem that density-dependent processes are minimized for these populations through urban drivers, such as an anthropogenic food supply that seems unending. However, as intuitive as this seems, the perception that density dependence is not as important in “overabundant” urban populations may be wrong. Density-dependent processes are notoriously difficult to document, and may be obfuscated by other factors that are more easily observed, especially when demographic processes that have been identified for a species in nonurban areas are severely altered in urban systems. More likely, density dependence for urban populations occur at different points in the trajectory of population growth, or primary regulatory mechanisms may switch with urbanization. The following examples illustrate the importance of density-dependent factors even for “overabundant” urban species.

For urban white-tailed deer at high population densities, food and water seem not as limiting as in nonurban sites as suggested by the high adult survival rates and fecundity of urban deer (Etter et al. 2002). An exception to the pattern is that canid predation on fawns appears to be density-dependent factor that that increases vulnerability of deer neonates. Through browsing abundant deer reduce the cover available for neonates, increase their predation risk and reducing fawn survival from 0.78 to 0.26 at densities of 100 adults/km^2 (Piccolo et al. 2010). Low fawn survival explains why deer populations at high densities in some urban greenspaces appear to experience no additional population growth, despite continued high adult survival and fecundity (Etter et al. 2002).

For other species not affected by predation, especially medium-sized mammals, disease may act in a density-dependent mechanism, such as sarcoptic mange. Sarcoptic mange often becomes epizootic at high host densities, and urban red fox populations may cycle as source and sink, depending on the prevalence of this disease (Gosselink et al. 2007). In peak years of the mange cycle, urban populations may serve as sinks, where mortality rates exceed reproduction and limited dispersal occurs, but during periods of low mange incidence survival in urban areas is relatively high and urban populations serve as sources with considerable juvenile dispersal into rural areas (Gosselink et al. 2007).

Although the abundant anthropogenic resources and reduced predation pressure may change when density-dependent factors manifest, many urban wildlife populations appear stable and experience some regulation such that they fluctuate less than rural populations (e.g., some rodent species, McCleery 2009; Chiappero et al. 2011). The apparent lack of an obvious density-dependent mechanism for a high-density species was evident in a raccoon population intensively monitored for 8 years, during which 647 individuals were captured 1452 times (Prange et al., 2003; Gehrt 2004). This population resided in an urban park with access to abundant and predictable sources of anthropogenic food, as the park received between 1.5 and 3 million human visitors annually (Gehrt 2004). Spring densities were quite consistent over the years whereas autumn densities fluctuated wildly (Fig. 8.7). The autumn densities were driven by juvenile recruitment, thus reflecting successful reproduction that year. Coefficient of variation for the autumn densities (48 %) was nearly four times that of spring densities (14 %), and there was clearly a disconnect between spring densities and the number of juveniles per adult female in the subsequent autumn ($r = -0.48$, $P = 0.28$). Nevertheless, some process occurred between autumn and the following spring (that is, over winter) that regulated the population. This process likely involved juveniles as they were not radiocollared, and probably involved over-winter survival or emigration, neither of which is mutually exclusive. Genetic evidence for this population, and others like it, indicated there was little immigration into the population (< 5 % of the study population originated outside the study area, Santonastaso et al. 2012), which was consistent with field data (Gehrt unpublished data), suggesting a general flow outward from the population and a possible density-dependent mechanism that is difficult to observe in this species.

Emigration is likely another important component of population regulation for urban adapter mammals (McCleery 2009; Soulsbury et al. 2010). Dispersal appears to be an important density-dependent mechanism in urban foxes (Gosselink 2002; Soulsbury et al. 2010), given their highly-structured social systems. Similarly, dispersal of young coyotes from urban populations with high survival and reproductive rates helps to maintain local densities at carrying capacity (Gehrt unpublished data).

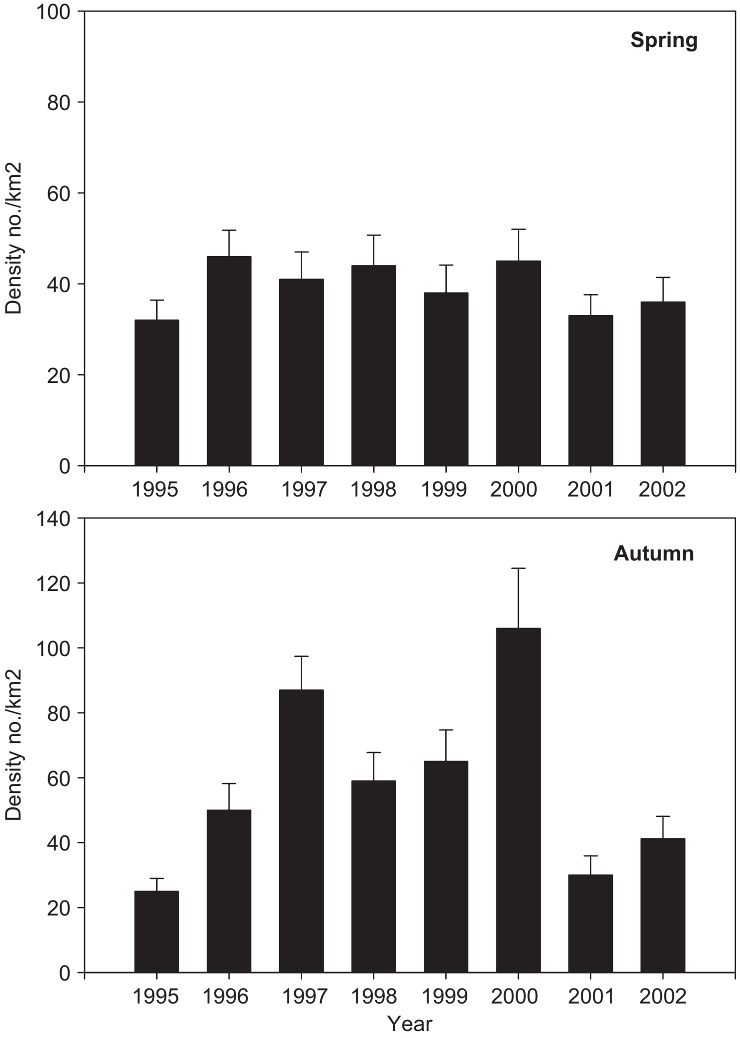


Fig. 8.7 Seasonal density estimates (+SE) for a raccoon population in an urban park with abundant and predictable anthropogenic foods in the Chicago metropolitan area, 1995–2002 (Gehrt 2004). Note the different scale on the y-axis. The population appeared to be at an artifical carrying capacity, as spring estimates exhibited little annual variability, whereas autumn estimates varied substantially across years

8.4 Metapopulations and Genetic Structure

Populations within metropolitan areas may occur as disjunct subpopulations with limited exchange of individuals. We would expect such a pattern to be especially extreme in terrestrial, sedentary species with specific habitat requirements.

Amphibians exhibit these characteristics (Hamer and McDonnell 2008), and genetic evidence suggests that they are susceptible to urban fragmentation of habitat, resulting in limited gene flow and reduction of genetic heterogeneity within subpopulations (Hitchings and Beebee 1997; Noël et al. 2007; Tsujia et al. 2011). For example, genetic differentiation of the common frog (*Rana temporaria*) between town ponds separated on average by only 2.3 km was nearly twice as high as that found for rural sites separated on average by 41 km (Hitchings and Beebee 1997). Similar genetic patterns occur in urban aquatic reptiles such as turtles (Rubina et al. 2001) and terrestrial lizards (Delaney et al. 2010).

Urban fragmentation of natural habitats also affects the dispersal abilities and genetic structure of small mammal populations (Gortata et al. 2012) and semi-fossorial medium-sized mammals (Huck et al. 2008). In a comparison of Sigmodontine rodent *Calomys musculinus* populations in an urban and agroecosystem environment in Argentina, urban subpopulations were genetically isolated and had higher level of relatedness than the subpopulations in the agricultural landscape (Chiappero et al. 2011). For species of small mammals that attain high densities in urban fragments such as the white-footed mouse (*Peromyscus leucopus*), genetic heterogeneity also occurs across subpopulations indicating limited dispersal between fragments, but genetic variability may remain relatively high within subpopulations because of high local densities (Munshi-South and Kharchenko 2010). As Munshi-South and Kharchenko (2010) assert, given that white-footed mice display genetic similarity over vast geographic regions outside the city, the substantial genetic structuring over short distances in the New York City system illustrates the power of urban processes even on “urban adapter” species.

It is perhaps intuitive that urbanization may fragment populations or limit dispersal of small-bodied species or those restricted to natural habitat fragments such as Eurasian badgers (Huck et al. 2008), but the strength of urbanization to fragment or structure populations is clear even when species with high mobility are considered. Genetic and behavioral data for foxes in Melbourne, Australia, and Zurich, Switzerland, revealed limited dispersal and relatively low genetic heterogeneity among urban foxes when compared to outlying rural populations (Robinson and Marks 2001; Wandeler et al. 2003). Extensive mark-recapture data across the urban landscape of Scarborough, Ontario, and recent genetic analysis of subpopulations across Chicago revealed metapopulation structure with limited dispersal and gene flow for raccoons, despite that species being the most abundant native mesomammal in both systems (Broadfoot et al. 2001; Santonastaso et al. 2012). However, gene flow was still sufficient to prevent loss of genetic diversity or drift due to limited dispersal in the Chicago system (Santonastaso et al. 2012). Similarly, genetic evidence suggests a limited, male-biased, dispersal among brush-tail possums in Australian cities (Stow et al. 2006). More surprising is when urbanization reduces population connectivity for birds such as wrentits (*Chamaea fasciata*; Delaney et al. 2010), and especially highly vagile avian species such as song sparrows (*Melospiza melodia*; Unfried et al. 2013) and house sparrows (*Passer domesticus*; Vangestel et al. 2012).

8.5 Conservation and Management Implications

Conservation and management of wildlife generally takes place at the population level, and this makes knowledge of population dynamics within cities important from a practical perspective. Understanding population dynamics of species negatively impacted by elements of urbanization is necessary to identify effective conservation activities (Hamer and McDonnell 2008), as well as to communicate to decision makers and the general public the need for controversial measures, such as removal of non-native vegetation from habitat fragments and control of domestic animals such as feral cats (Chap. 11). Knowledge of demographic parameters is often essential for effective management of “overabundant” urban wildlife. For example, deer (*Odocoileus* sp.) are typically the largest vertebrates inhabiting urbanized landscapes across North America. Consequently, they are quite obvious to the public, providing viewing opportunities, while at the same time causing substantial property damage and collisions with vehicles (Warren 2011). In particular, the proliferation of white-tailed deer in urban parks and residential areas has presented management challenges (Chap. 17). As a result, municipalities and management agencies expend considerable effort and expense to estimate population demographics of deer populations to determine acceptable population levels and to provide support for deer management to the public or decision makers (Jones and Witham 1995; De Nicola et al. 2000; LaBonte et al. 2004; DeNicola et al. 2008). Reliable estimates of population density and demographic structure are critical for identifying management goals such as harvest numbers for culling programs or for justifying such management to the public (LaBonte and Barclay 2007).

Understanding population dynamics of urban wildlife also has implications for human health, which is increasingly subject to emerging zoonoses. In this case, demographic information and dispersal patterns are instrumental for the management of certain wildlife diseases, such as rabies and canine distemper (Rosatte et al. 2007a). Rabies management typically involves culling or baiting programs (Rosatte et al. 2007b), and density estimates are necessary to determine the number of animals to remove and the number of vaccine baits to distribute (Rosatte et al. 1992, 1997).

Conclusion

We extol the value of explicitly considering the population ecology of urban wildlife, yet we recognize that demographic studies are time- and resource-intensive and, hence, not always possible. Application of general rules of thumb can sometimes be a reasonable surrogate for place-based studies, but this may prove difficult within urbanizing systems. One of the most striking patterns to emerge from our review was the absence of any clear “rules” governing population ecology of urban wildlife. Though there are several common patterns in density and/or demography, sufficient exceptions exist to preclude widespread generalization. This fact at once makes urban wildlife populations especially interesting from a scientific perspective and especially challenging from a conservation and management one.

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Chapter 9
Urban Wildlife Behavior

Amy M. Ryan and Sarah R. Partan

Keywords Animal communication · Anti-predator behavior · Behavioral plasticity · Foraging · Urban noise · Urbanization

9.1 Introduction

Wild animals in urban areas face unique challenges. They live in environments modified by and for humans without having evolved in them, but unlike domesticated animals, they remain under the auspices of natural selection. We do not yet know how living in urban areas will ultimately affect the animals that share our urban environments, but we are at the beginning of a new and important scientific effort to study the effects of urbanization on wildlife. Early signs suggest that animals that can tolerate urbanization are quite different in behavior and physiology than those that are limited in distribution to natural areas (McCleery et al. 2011).

In this chapter, we focus on how animal behavior changes in response to urbanization. We use the term *urban* to refer to habitats undergoing intense human development for purposes of human residency, as seen in cities and suburbs (see Gehrt 2010). Understanding why some animals successfully adapt their behavior to urban areas and others do not can contribute to both the management of urban wildlife and the conservation of species that do not fare well during urbanization. After addressing the behavioral effects of urbanization, we conclude with implications for evolution and biodiversity of animals in urban environments. We have limited our review to terrestrial vertebrates with a few exceptions. Aquatic animal

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S. R. Partan (✉)
School of Cognitive Science, Hampshire College, Amherst, MA, USA
e-mail: partan@hampshire.edu

A. M. Ryan
Neuroscience and Behavior Program, University of Massachusetts Amherst,
Amherst, MA, USA
e-mail: amyr@cns.umass.edu

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behavioral responses to urbanization are reviewed by Candolin (2009), Slabbekoorn et al. (2010), and van der Sluijs et al. (2011).

9.2 Animal Movement and Home Ranges

An animal's range of movements in the urban environment reflects the costs and benefits of factors such as clumped resources, high density of conspecifics, and anthropogenic barriers to movement. The urban landscape can be characterized as fragments of suitable habitat surrounded by unusable areas that animals must travel around or through to reach other habitat fragments. In contrast to natural areas, urban habitat fragments, including parks, residential yards, and other green spaces, often provide highly clumped resources (e.g., plants, anthropogenic food, shelter, and prey) but few resources exist between fragments.

Across taxa, animals in urban areas tend to have smaller home ranges compared with conspecifics in nonurban areas (Etter et al. 2002; Prange et al. 2004; Hoffman and O'Riain 2011). Large carnivores are sometimes an exception, with no consistent changes in home range size associated with urbanization, perhaps because urban habitat quality is inconsistent. Urban bobcat (*Lynx rufus*) populations have exhibited both larger and smaller home ranges than their more rural counterparts (Tigas et al. 2002; Riley et al. 2003), but urban black bears (*Ursus americanus*) exhibited smaller home ranges than rural populations (Beckmann and Berger 2003a), and urban coyotes (*Canis latrans*) exhibited larger home ranges than adjacent rural populations (Grinder and Krausman 2001; Riley et al. 2003). For most animals, however, the smaller home ranges observed in urban areas lead to higher densities, which has implications for territorial and social behavior (see Sect. 9.6 below) and can affect dispersal dynamics (see Chap. 8).

Clumped anthropogenic food resources have a strong influence on home range sizes. In some cases, food is so abundant that animals can reduce travel time and energy required to forage. For example, baboons (*Papio ursinus*) with access to anthropogenic food sources near human settlements traveled less on a daily basis than baboons in more natural areas (Hoffman and O'Riain 2011), and urban raccoons (*Procyon lotor*) had smaller home ranges compared with rural raccoons (Prange et al. 2004; Adams and Lindsey 2010). Anthropogenic food resources also tend to be available year-round, so animals may not need to alter their home range in order to adjust for seasonal variations in food sources. Thus, in addition to reduced size, urban animals can have comparatively stable home ranges across seasons (Prange et al. 2004).

Roads are a pervasive source of fragmentation; it was estimated in 1998 that about 1 % of the USA was covered in roads (Forman and Alexander 1998). Because roads trigger animal avoidance behavior, they can create a barrier for movement. Jaeger et al. (2005) suggested that differing responses to roads can be seen in the avoidance of traffic noise, road surface, or vehicles. Taxonomic differences in responses to these three different disturbances may account for different responses to roads observed across taxa. In terms of traffic noise, birds and anurans that rely heavily on acoustic communication have been detected in lower densities near roads than in other areas (birds: Reijnen et al. 1995; Rheindt 2003; Goodwin and Shriver 2010; anurans: Fahrig et al. 1995), perhaps because the noise pollution negatively affects their ability to communicate. However, these effects are difficult to distinguish from the need to avoid roads due to vehicle avoidance or habitat modification. Noise pollution and its effects on animal communication will be discussed further in Sect. 9.6.2.

Road surface avoidance describes wildlife that do not cross roads regardless of the presence of vehicular traffic loads. Some animals do not extend their range to traverse roads or buildings, suggesting that roads and buildings can serve to define home ranges and create linear range borders (Leader et al. 2000; Etter et al. 2002; McCleery and Parker 2011). The avoidance of vehicular traffic differs from road surface avoidance in that some animals may cross roads but appear to cross only when traffic is decreased, such as nighttime (Jaeger et al. 2005). This has been observed among mammals as small as hedgehogs (*Erinaceus europaeus*) (Dowding et al. 2010) and as large as moose (*Alces alces*) (Eldegard et al. 2012).

Finally, some animals display maladaptive behaviors on roads or near approaching vehicles that lead to mortality. For example, ectothermic animals such as reptiles may be attracted to warm road surfaces for thermoregulation, putting them at risk (MacKinnon et al. 2005). Other maladaptive behaviors include animals freezing in place in front of approaching cars, which is a common behavior among animals that are otherwise not exposed to as intense light as that from headlights (Mazerolle et al. 2005). In a particularly maladaptive response to vehicles, armadillos (*Dasypus novemcinctus*) often jump straight up when a car passes over them (Inbar and Mayer 1999). This behavior possibly evolved to startle predators when armadillos perceive a threat, but in the case of cars, usually results in mortality (Inbar and Mayer 1999). Amphibians are especially vulnerable to mortality from road crossings because they require both aquatic and terrestrial habitats and may need to cross roads to reach these different habitats (Fahrig et al. 1995; Hels and Buchwald 2001; Hamer and McDonnell 2008; Rytwinski and Fahrig 2012).

9.3 Use of Human Structures

Urban adapters, particularly birds and small mammals, often utilize human-made structures for shelters, nests, or dens for raising and protecting offspring (Fig. 9.1). For example, over 50 % of raccoon dens occur in human-made structures (Hoffmann and Gottschang 1977; O'Donnell and DeNicola 2006), such as attics, chimneys, garages, sheds, ventilation ducts, furniture (Adams and Lindsey 2010), and sewers (Hoffmann and Gottschang 1977). Other mammals that use human-made structures are striped field mice (*Apodemus agrarius*) (Gliwicz et al. 1994), fox squirrels (*Sciurus niger*) (McCleery et al. 2007), stone martens (*Martes foina*) (Herr et al. 2010), skunks (*Mephitis* spp.) (Rosatte et al. 2010), and red foxes (*Vulpes vulpes*), which appear to be less tolerant of human presence and den mainly in abandoned

Fig. 9.1 Robin nesting on a porch in a town in Western Massachusetts. (Photograph by SRP)

structures (Soulsbury et al. 2010). Many species of birds construct nests on human-made structures and some utilize artificial objects such as yarn or foil in their nests. In Chinese bulbuls (*Pycnonotus sinensis*), there was a positive correlation between the degree of urbanization and the amount of anthropogenic material used in nests (Wang et al. 2009). Some anthropogenic items found in nests may serve other functions in addition to improving nest structure. For example, cigarette butts utilized in nest-building appear to have anti-parasitic effects on the nest (Suárez-Rodríguez et al. 2012).

Bridges provide a rather unique opportunity for protected roosts because they are elevated, offering protection from predators, and provide warmth due to the heat-retention properties of concrete. Bats utilize bridges for roosting (Lewis 1994; McCleery 2010), sometimes in large numbers; for example, 1.5 million Mexican free-tailed bats (*Tadarida brasiliensis*) roost under the Congress Avenue Bridge in Austin, Texas (Adams and Lindsey 2010). Birds that successfully utilize bridges for nesting tend to be cliff or cavity nesters such as swifts (*Apodidae* family) and swallows (*Hirundinidae* family) (Marzluff 2001).

The propensity to live in human structures probably depends both on the shelter requirements for the species and an individual's tolerance for human activity near

Fig. 9.2 Herring gull foraging from a trash can on Cape Cod, Massachusetts. (Photograph by Dennis Dietz)

its shelter. Human-made structures have many benefits for animals, but can also be a major source of mortality for animals of flight. Although birds can deftly navigate through a forest of trees, about 550 million birds die each year in the USA from collisions with structures built by humans, especially buildings with large glass windows (Erickson et al. 2005), perhaps due to the illusion of open space created by transparent or reflective glass and/or disorientation from light reflecting off the glass (Klem 1989).

9.4 Foraging Behavior

The foraging behaviors of urban animals are likely best understood through the lens of the profoundly altered urban food web. Humans help shape the urban food web by introducing species, both directly (e.g., domesticated cats (*Felis catus*) and dogs (*Canis familiaris*)) and indirectly (e.g., urban exploiters such as rock doves (*Columba livia*), and Norway rats (*Rattus norvegicus*)) (Faeth et al. 2005; McKinney 2006). At the top of the food web, urbanization tends to remove many large predators, facilitating the success of medium-sized predators, or *mesopredators*, including red foxes, raccoons, and coyotes (Crooks et al. 2010; Bateman and Fleming 2012). These patterns coupled with the urban environment's selection for generalist and edge species creates a rather homogenous group of fauna in the urban food web (Chap. 7).

At the base of the urban food web, food availability, distribution, and nutritional characteristics are profoundly altered by the infusion of anthropogenic sources of food. Human settlements provide food to urban wildlife via garbage, non-native plants, pet food, road kill, and intentional feeding such as birdseed (Chaps. 4, 6; Warren et al. 2006a). Of these food sources, garbage is the predominant source of food for most urban mammals and some birds (Fig. 9.2) (Belant et al. 1998; Beckmann and Berger 2003b). Much of the waste generated by humans is never completely removed from the environment but placed in progressively more concentrated refuse areas, allowing continued access by animals. Overall,

Table 9.1 Percentage of anthropogenic foods incorporated into the diets of urban animal populations

Species	Percentage of Diet that Consists of Anthropogenic Food (%)	Method of Measurement	References
Ring-billed gull (*Larus delawarensis*)	54	Analyzed contents of pellets	Belant et al. (1998)
Red fox (*Vulpes vulpes*)	>50	Analyzed stomach contents of deceased foxes	Contesse et al. (2004)
Baboon (*Papio cynocephalus*)	>38	Observed behavior	Altmann and Muruthi (1998). Note: Not urban, but the group had access to garbage dump
Eastern gray squirrel (*Sciurus carolinensis*)	35	Observed behavior	Parker and Nilon (2008)
Herring gull (*Larus argentatus*)	20	Analyzed contents of pellets	Belant et al. (1998)
Florida scrub jay (*Aphelocoma coerulescens*)	15	Observed behavior	Fleischer et al. (2003)
Coyote (*Canis latrans*)	14–25	Analyzed contents of fecal samples	Fedriani et al. (2001)

anthropogenic sources of food are nutritionally and calorically rich, highly concentrated in distribution, replenished after depletion, and consistently provided over time (Beckmann and Berger 2003b; Prange et al. 2004). Animals that incorporate the most anthropogenic food in their diets tend to be generalists or omnivores (McKinney 2002, McCleery 2010). The percentage of anthropogenic foods in urban wildlife diets is reported in Table 9.1, and additional carnivore diets are outlined by Iossa et al. (2010).

The addition of anthropogenic foods to the diets of urban wildlife affects their behavior. Many of these effects are inexorably linked with those of urbanization in general, such as smaller home ranges and increased aggregation, but the highly clumped and nutritious sources of food also decrease time spent foraging. Monkeys in or near urban areas are particularly opportunistic in pursuing anthropogenic food. Urban monkeys in Kenya and Uganda spent less time foraging and more time resting and socializing than did their conspecifics in more natural areas (Altmann and Muruthi 1988; Saj et al. 1999, respectively). For example, baboons (*Papio cynocephalus*) living near a local garbage dump spent 20% of their time feeding and 50% of their time resting; in contrast, a nearby group relying on natural food sources spent approximately 60% of their time feeding and only 10% of their time resting (Altmann and Muruthi 1988). A difference in activity budgets was also found in Florida scrub-jays (*Aphelocoma coerulescens*), where females in suburban areas foraged 12% less and perched 11% more of the time than did females in more natural areas (Fleischer et al. 2003).

In addition to spending less time foraging, urban wildlife may be more efficient foragers. The suburban Florida scrub-jays that spent less time foraging also consumed more food per hour than scrub-jays in more natural areas (Fleischer et al. 2003). Foraging efficiency has been experimentally measured as the giving-up density (GUD), which is the density of food remaining in an artificial patch when an animal ceases foraging (Bowers and Breland 1996). How long an animal spends at a food patch, measured by how much food is left when it leaves, quantifies the animal's assessment of the trade-off between foraging cost and predation risk (Shochat et al. 2004). Eastern gray squirrels (*Sciurus carolinensis*) in more urban areas had greater densities and lower GUDs than squirrels in more rural areas (Bowers and Breland 1996). Similarly, birds in urban areas have shown consistently lower GUDs than birds in nonurban areas (Shochat et al. 2004; Lerman et al. 2012).

There are multiple explanations for why urban animals may have lower GUDs than nonurban ones. Urban animals may perceive less predation pressure because of the altered community structure in urban areas and thus enjoy longer foraging bouts (Bowers and Breland 1996; Shochat et al. 2004; Lerman et al. 2012), or they may be less cautious because urban areas select for bold personalities, as discussed in Sect. 9.7.1. However, a comparison of birds in native desert (xeric) yards and in non-native lush (mesic) yards showed no differences in perceived predation risk as expressed through the use of foraging trays placed closer or farther from protective cover (Lerman et al. 2012). There was a lower GUD in mesic yards, parallel to the lower GUD of urban areas, which may be explained by a greater density of animals living in mesic yards than xeric yards (Lerman et al. 2012). Lower predation and abundant resources may contribute to a greater density of conspecifics in urban areas as well, leading to greater competition for food and longer foraging bouts to procure enough calories (Bowers and Breland 1996; Shochat et al. 2004; Lerman et al. 2012).

In sum, because urban areas have a profusion of anthropogenic structures and sources of food, those animals with the behavioral flexibility allowing them to take advantage of these resources should fare best (behavioral flexibility will be discussed in Sect. 9.7.1 below).

9.5 Anti-predator Behavior and Response to Humans

The continual presence of humans in the urban environment creates a novel challenge for urban animals in part because most wildlife species have evolved to view humans as a competitor or a predator. As such, taxa in urban areas are commonly disturbed by humans even when there is little risk of harm (Frid and Dill 2002). The repeated exposure to humans in urban settings may disrupt animal activities and cause wildlife to expend energy in vigilance or avoidance behavior.

For prey animals, avoidance and vigilance behaviors come at the cost of time spent in other survival-promoting activities such as foraging. Although disturbance from humans is not as likely to be lethal as is disturbance from other predators, prey animals that encounter humans often give up foraging time for vigilance (Lima and Dill 1990; Frid and Dill 2002). The decision to stop foraging and become vigilant or flee is influenced by factors such as group size and how close the prey animal is to protective cover. Whether or not the perceived predator ultimately attacks can

Table 9.2 Summary of studies that have compared flight initiation distance (FID) measures in conspecifics from urban and nonurban areas

Species	Did urban animals have shorter FIDs than nonurban animals?	References
Birds		
Crow (*Corvus brachyrhynchos*)	Yes	Knight et al. (1987)
Black-billed magpie (*Pica pica*)	Yes	Fernández-Juricic et al. (2001); Jerzak (2001)
Common wood pigeon (*Columba palumbus*)	Yes	Fernández-Juricic et al. (2001)
European blackbird (*Turdus merula*)	Yes	Fernández-Juricic et al. (2001)
House sparrow (*Passer domesticus*)	Yes	Fernández-Juricic et al. (2001)
Western Gulls (*Larus occidentalis*)	Yes	Webb and Blumstein (2005)
44 species in Europe	Yes	Møller (2008)
House finch (*Carpodacus mexicanus*)	No	Valcarcel and Fernández-Juricic (2009)
42 species in South America	Yes	Carrete and Tella (2011)
Mammals		
Black-tailed prairie dogs (*Cynomys ludovicianus*)	Yes	Magle et al. (2005)
Eastern grey squirrel (*Sciurus carolinensis*)	Yes	Cooper et al. (2008); Parker and Nilon (2008)
Fox squirrel (*Sciurus niger*)	Yes	McCleery (2009)

thus be irrelevant to the prey's decision as long as the predator stimulus is initially perceived as a threat (Frid and Dill 2002). For example, bighorn sheep (*Ovis canadensis*) decreased foraging and resting and increased vigilance when aircraft flew overhead (Stockwell et al. 1991); pink-footed geese (*Anser brachyrhynchus*) spent less time feeding and consumed less food in fields with high levels of anthropogenic disturbance than in more remote areas (Gill et al. 1996); and sanderling (*Calidris alba*) foraging bouts decreased as the number of people on their beach increased (Thomas et al. 2003).

The effect of human disturbance on animal behavior has been assessed experimentally using a measure of flight initiation distance (FID). In this technique, a human directly approaches an animal until the animal alters its behavior by fleeing; the distance at which its behavior changes indicates its disturbance threshold (Frid and Dill 2002). How closely a human can approach an animal is thus indicative of its perceived predation risk and tolerance for humans. In general, when FIDs are compared between conspecifics in urban and nonurban environments, the urban animals have shorter FIDs, allowing humans to approach them at closer distances, than rural ones (Table 9.2). This may be because urban prey species have learned that humans that maintain a minimal distance do not usually prey on animals and are ultimately benign (Frid and Dill 2002; McCleery 2009). However, this phenomenon may not apply to anurans, for their FID seems to remain consistent with no signs of habituation to humans (Rodríguez-Prieto and Fernández-Juricic 2005).

Aside from FID measures, urban animals show other indications of being less wary around humans than do conspecifics in less urbanized landscapes. For example, escape behaviors exhibited during handling by humans were found to be different between urban and rural individuals of 15 species of birds in Europe, with urban birds exhibiting less intense physical escape behaviors than rural birds (Møller and Ibáñez-Álamo 2012). Additionally, when tested for temperament, eastern chipmunks (*Tamias striatus*) that resided in areas used frequently by humans were more docile and explorative than chipmunks that lived in the same nature reserve but in areas with less human activity (Martin and Réale 2008). These results align with Carrete and Tella's (2011) finding that across 42 South American bird species, birds found in urban areas were more tolerant of humans than were conspecifics in nonurban areas. Collectively, these results suggest the phenotypic trait of "tameness" was more prevalent in animals that lived in areas of great human activity.

Engaging in anti-predator behavior can come at a cost to parental behavior, which has the potential to create direct fitness consequences for animals that do not tolerate repeated exposure to humans. For example, with exposure to fishermen, marsh harriers (*Circus aeruginosus*) decreased parental behavior, increased alarm behavior, and ultimately had less healthy chicks (Fernández and Azkona 1993).

Both predators and prey that occur in urban areas can shift the timing or location of their activities to avoid high levels of human activity. Predators in urbanized areas tend to shift their traveling and hunting activity to nighttime, when human activity decreases (black bears: Beckmann and Berger 2003b; javelinas [*Tayassuta jacu*]: Ticer et al. 1998; coyotes: Grinder and Krausman 2001; Riley et al. 2003, but see Tigas et al. 2002). In less urban areas, these predators are more active at dawn and dusk than at night. When urban predators alter their activity times based on human rather than prey activity, they may lose opportunities to hunt preferred prey that are not nocturnal (Ditchkoff et al. 2006), but gain opportunities to target new species of prey or to become increasingly omnivorous. Similarly, hedgehogs in residential areas shifted their activities to later hours at night when humans and their pets were not present (Dowding et al. 2010) and marmosets (*Callithrix penicillata*) in urban parks shifted their activities spatially and moved to areas further from human activity (Duarte et al. 2011) and inaccessible to feral cats (Duarte and Young 2011). Additionally, frogs (*Rana iberica*) decreased their use of stream banks in areas of human recreational activity (Rodríguez-Prieto and Fernández-Juricic 2005). Shifts in activity can isolate these individuals from preferred food and shelter if animals move to lower quality habitat in efforts to avoid humans (Frid and Dill 2002; Duarte et al. 2011).

It is possible that individuals with reduced responses to environmental stressors can exploit the resources of urban environments that more sensitive conspecifics cannot tolerate. However, it is currently unclear whether individual animals in urban environments are more tolerant of humans due to habituation (McCleery 2009) or whether there is a selection bias toward animals that are more tolerant or docile (Martin and Réale 2008). It may be that the most predictive factor explaining an individual's increased tolerance to humans is how many generations the species has been present in urban areas (Møller 2008). The mechanism behind tolerance for humans may not be the same for all species; however, a consistent trait of prey species that successfully colonize urban areas is that they respond adaptively to the increase in human activity around them.

9.6 Social Behavior

Whether animals aggregate in groups or remain solitary, they must organize themselves within a landscape and interact with conspecifics at least occasionally. Animals that are considered solitary are often territorial and consequently have a low tolerance for conspecifics in terms of sharing resources or space. In contrast, group-living animals have a greater tolerance for sharing resources or space with other group members although their interactions may be modulated by social hierarchies within the group. The effects of urbanization on social behavior have not been as well studied as have the effects of urbanization on other animal behaviors (Yahner and Mahan 1997; Banks et al. 2007), but social dynamics are expected to be altered by changes in conspecific and predator density as well as the availablility of food and shelter.

9.6.1 Territoriality

The higher density of conspecifics common in urban environments (Prange et al. 2003; Banks et al. 2007; McCleery 2010) affects social organization. In particular, territorial animals in fragmented urban landscapes are faced with the risk of either losing territory because of the high density of conspecifics, or abandoning their territorial organization altogether and tolerating aggregations of individuals. It is also possible that the structure of urban groups does not diverge from their structure in more natural areas but is simply more apparent in urban areas where animal density is highest (Prange et al. 2011).

There is likely to be increased competition for concentrated resources as more animals aggregate in an area and direct access to food decreases for each individual (Wrangham 1974; Shochat 2004). Animals in higher densities may thus become territorial and increase aggressive behavior to defend their portion of resources in the fragment (Wiens 1976). For example, urban fox squirrels were found to have smaller home ranges with less range overlap and used fewer nests than rural squirrels (McCleery and Parker 2011). Because dead trees are typically removed from urban areas, there may have been fewer tree cavities available for urban squirrels. With a limited supply of nesting sites, squirrels then used sites exclusively and defended them against other squirrels (McCleery and Parker 2011). Urban gray squirrels also showed an increase in aggressive behavior with greater density (Parker and

Nilon 2008). Among birds, both density and intraspecific aggression increased in urban blackbirds (*Turdus merula*) (Gliwicz et al. 1994), and territorial behavior was greater in urban than desert towhees (*Melozone aberti*) and curve-billed thrashers (*Toxostoma curvirostre*), although this difference was not related to bird density in the thrashers (Fokidis et al. 2011). As Fokidis et al. (2011) suggested, there can be other habitat or physiological factors such as resource availability and hormonal processes that affect territorial behavior in urban animals.

Other urban animals, however, respond to urban conditions by relaxing territorial behavior, tolerating other individuals, and aggregating into large, non-territorial groups. This is most frequently observed in what are usually considered solitary carnivores such as foxes (Macdonald et al. 1999), raccoons (Prange et al. 2004), southern hairy-nosed wombats (*Lasiorhinus latifrons*) (Walker et al. 2008), and feral cats (Baker et al. 2010). Less territorial behavior was also observed in urban badgers (*Meles meles*) as compared to rural badgers (Harris et al. 2010; Bateman and Fleming 2012), although badgers living in groups can experience high intraspecific aggression (Macdonald et al. 2004).

Most studies of urban animal sociality focus on large-scale differences in social behavior between urban and rural environments, rather than on behavioral differences among individuals within these environments. There can be great intraspecific variability in social behavior, however, making it difficult to generalize behaviors observed in one group to other groups (Banks et al. 2007). Furthermore, intraspecific interactions play an important role in mate access and mate choice, so more investigation is needed into how individual behavior within a population is affected by urban environments.

9.6.2 Animal Communication

Wildlife communication in urban areas represents a rapidly growing discipline (e.g., Slabbekoorn and Peet 2003; see Partan 2013). Because the environment is the medium for signal transfer, it plays a large role in the evolution of animal communication signals. Signal evolution is influenced by the sending and receiving capabilities of sensory systems and by the environment through which the signal must travel (Endler 1992). Effective signals must penetrate background environmental noise with minimal degradation to reach recipients; this becomes challenging in urban settings due to increased acoustic noise (e.g., traffic sounds), chemical noise (e.g., vehicle emissions), and visual noise (e.g., light pollution). Environmental change such as urbanization that presents animals with these novel sensory conditions can therefore have the potential to drive evolution.

The environmental changes that occur during urbanization provide an opportunity to observe how species are affected by alterations in the sensory environment and to assess whether individuals or populations can modify their communication signals to meet the challenges presented by urban environments (Table 9.3). The acoustic environment has been the most frequently studied in this regard; urban

Table 9.3 Acoustic signal modifications documented in urban areas as compared with natural areas

Type of Signal Modification	Species	References
Frequency (pitch)		
Increase minimum frequency	great tit (*Parus major*)	Slabbekoorn and Peet (2003); Slabbekoorn and den Boer-Visser (2006); Mockford and Marshall (2009)
	house finch (*Carpodacus mexicanus)*	Fernández-Juricic et al. (2005)
	song sparrow (*Melospiza melodia*)	Wood and Yezerinac (2006)
	European blackbird (*Turdus merula*)	Ripmeester et al. (2010); Hu and Cardoso (2010); Mendes et al. (2011)
	rainbow lorikeet (*Trichoglossus haematodus*)	Hu and Cardoso (2010)
	Eastern rosella (*Platycercus eximius*)	Hu and Cardoso (2010)
	red wattlebird (*Anthochaera carunculata*)	Hu and Cardoso (2010)
	bell miner (*Manorina melanophrys*)	Hu and Cardoso (2010)
	gray catbird (*Dumetella carolinensis*)	Dowling et al. (2011)
	Northern cardinal (*Cardinalis cardinalis*)	Dowling et al. (2011)
Increase maximum frequency	European blackbird (*Turdus merula*)	Ripmeester et al. (2010); Mendes et al. (2011)
Increase dominant frequency	green frog (*Rana clamitans*)	Cunnington and Fahrig (2010)
	Northern leopard frog (*Rana pipiens*)	Cunnington and Fahrig (2010)
Shift the emphasis of calls from lower to higher harmonics	California ground squirrel (*Spermophilus beecheyi*)	Rabin et al. (2003)
Duration		
Increase duration of calls once noise exceeds a threshold level	orca (*Orcinus orca*)	Foote et al. (2004[a])
Amplitude		
Increase amplitude of vocalizations in noise—known as the Lombard effect	common nightingale (*Luscinia megarhynchos*)	Brumm (2004)
Decrease amplitude of vocalizations	green frog (*Rana clamitans*)	Cunnington and Fahrig (2010)
Rate of occurrence		
Increase number of vocalizations	Taipei frog (*Rana taipehensis*)	Sun and Narins (2005[b])
	Eastern gray squirrel (*Sciurus carolinensis*)	Partan et al. (2010)
	Serins (*Serinus serinus*)	Díaz et al. (2011)

Table 9.3 (continued)

Type of Signal Modification	Species	References
Decrease number of vocalizations	painted chorus frog (*Microhyla butleri*)	Sun and Narins (2005[b])
	sapgreen stream frog (*Rana nigrovittata*)	Sun and Narins (2005[b])
	tree frog (*Hyla arborea*)	Lengagne (2008[c])
	green frog (*Rana clamitans*)	Cunnington and Fahrig (2010)
	Northern leopard frog (*Rana pipiens*)	Cunnington and Fahrig (2010)
	gray treefrog (*Hyla versicolor*)	Cunnington and Fahrig (2010)

[a] Asssessed from same population of whales during more noisy and less noisy conditions
[b] Assessed from same population of frogs during more noisy and less noisy conditions
[c] Determined via experimental playback of traffic noise

animals are confronted with increased ambient noise, increased spatial variations in noise level, and sound reflection and attenuation generated from vertical flat and impervious surfaces (Rabin and Greene 2002; Warren et al. 2006b; Mockford et al. 2011). Vehicular and air traffic, electricity, and electrical devices tend to be loud and low in frequency (Slabbekoorn and Peet 2003; Warren et al. 2006b), masking low-frequency animal signals. In addition to masking, sustained loud noise can lead to hearing loss in animals (Rabin et al. 2003) as well as other damaging physiological effects (Kight and Swaddle 2011), profoundly affecting their ability to communicate successfully.

The effects of urbanization on acoustic communication are studied most in relation to songbirds (passerines). Songbirds use auditory communication for functions such as establishing territory, advertising to and assessing mates, alarm calling at predators, and parental care through begging calls made by nestlings (Patricelli and Blickley 2006; Slabbekoorn and Halfwerk 2009). Songbirds are capable of vocal learning and can fine-tune their song structure with specialized syrinx muscles, so they are considered to be behaviorally flexible and able to alter their songs to suit a changing environment (Patricelli and Blickley 2006; Cardoso and Atwell 2011). However, it is evident that not all species of songbirds fare well in urban areas; the differences in success among species suggest that flexibility to adjust to an acoustically altered environment can be limited (Warren et al. 2006b; Slabbekoorn and Ripmeester 2008; Halfwerk et al. 2011a).

Urban songbirds can avoid acoustic masking of their song by dispersing to less noisy environments, altering the timing of songs, or changing the acoustic structure of songs. Birds that sing low-frequency songs that overlap in frequency with urban noise sources may choose not to live in noisy areas. Certain species avoid living near such concentrated points of low-frequency noise as roads, and species richness near roads tends to be lower overall when compared with areas farther from roads (Reijnen et al. 1995; Rheindt 2003; Goodwin and Shriver 2010).

Birds can alter the timing of their songs based on temporal variation in ambient noise. Urban birds such as robins (*Erithacus rubecula*) have been observed singing earlier in the day than dawn; while this timing shift has been linked to the timing of anthropogenic noise (Fuller et al. 2007), the increase in artificial light also may affect the timing of morning song (Miller 2006).

When birds compete directly with low-frequency anthropogenic sources of noise, they can adjust the amplitude or structure of their song to avoid acoustic masking (see Table 9.3). Increasing amplitude to overcome background noise is referred to as the Lombard effect, and has been observed in common nightingales (*Luscinia megarhynchos,* Brumm 2004). Another strategy employed is to raise the minimum frequency of songs, although it is currently unclear whether this is a separate strategy or a consequence of the Lombard effect (Brumm and Zollinger 2011; Cardoso and Atwell 2011; Zollinger et al. 2012). When rural and urban conspecific birdsong is compared (examples in Table 9.3), the songs are generally very similar, often using the same dominant frequency, but the minimum frequency produced is higher in the urban than in the nonurban song. Non-passerines, such as parrots, also increase the minimum frequency of vocalizations in urban areas (Hu and Cardoso 2010).

The acoustic adjustments to urban birdsong may have functional consequences for birds. Song, generally produced by males, functions in courtship and territoriality. Scientists have attempted to gauge what effects altered song has on the receiver and, ultimately, female mate preference. Among great tits (*Parus major*) in both laboratory and field settings, signal detection through noise is affected by frequency, such that songs with a higher minimum frequency are easier to detect in noisier areas than songs with a lower minimum frequency. In the lab, higher-frequency songs were easier to detect than lower frequency songs through playbacks of background urban noise, but there was no difference in detection ability through playbacks of forest noise (Pohl et al. 2012). Concordantly, in the field setting, territorial males in urban environments responded most strongly to songs with higher minimum frequency during playback experiments (Mockford and Marshall 2009; Ripmeester et al. 2010). However, when song signal transmission and degradation are explicitly measured, a higher minimum frequency of songs only moderately improves signal transmission in urban environments (Nemeth and Brumm 2010; Mockford et al. 2011). Birds thus may make other adjustments to their song (e.g., increasing amplitude) to compensate for acoustic masking (Nemeth and Brumm 2010; Mockford et al. 2011). The relationship between the increased ambient noise in urban areas and change in bird song structure is complex, with multiple factors involved in how birds in urban areas adjust when, where, and how they communicate.

With a change in the structure of vocalizations in urban areas, there is potential for divergence in communication behavior between urban and rural conspecifics. For example, territorial male urban European blackbirds responded more strongly to higher-frequency urban intruder songs than to rural songs, whereas those in rural areas responded more strongly to the rural and lower-frequency intruder songs (Ripmeester et al. 2010). There may be a disconnect between songs best detected in the urban habitat and female song preference, given that the latter was selected over generations of time living in natural areas. In urban great tits, females responded to songs of high frequency in both noisy and control conditions but responded to low-frequency songs less in the noisy condition than in the control (Halfwerk et al. 2011b). However, it has been found that males with low-frequency songs seem to be cuckolded less often (Halfwerk et al. 2011b). Females thus may not hear their preferred males as well in urban areas. The change in communication behavior in urban birds can potentially isolate urban and rural populations if the change alters such factors as female preference, male attractiveness, how individuals recognize one another, and overall fitness costs to acoustic communication (Boughman 2002; Patricelli and Blickley 2006; Halfwerk et al. 2011b).

Two other taxa that have been studied for the effect of anthropogenic noise on acoustic communication are frogs and squirrels. These animals do not have the same dispersal opportunities or vocal flexibility as birds but nonetheless appear to devise ways to adjust their signals for improved signal transmission through noise. Frogs rely heavily on acoustic communication during the mating season with male advertisement calls influencing female preference for particular males (Bee and Swanson 2007). Playback experiments in the laboratory demonstrated that traffic noise can mask male calls and that female gray treefrogs (*Hyla chrysoscelis*) had both a longer response latency and a less intense response to calls that were masked by traffic noise than to those that were not (Bee and Swanson 2007). The masking of frog calls observed in the laboratory setting may help explain results observed in the field, where several species of frogs in Thailand and France changed their call bout duration during traffic playbacks (Sun and Narins 2005; Lengagne 2008). It appears that frog calls are affected by anthropogenic noise and that many frogs alter the timing of when and how much they call when exposed to urban noises, presumably to improve sound detection by receivers. In addition, recent evidence suggests that the vocalizations of frogs in noisy areas may be different than those in less noisy areas in terms of pitch (Parris et al. 2009; Cunnington and Fahrig 2010), and amplitude (Cunnington and Fahrig 2010). Frogs may, therefore, be able to adjust the actual structure of their calls when in areas affected by anthropogenic noise. However, the potential for vocal flexibility in anurans is in need of further investigation.

Squirrels and other mammals typically use vocalizations for functions other than mate attraction or territorial calls, such as social behavior or anti-predator behavior. As opposed to the advertisement function of many bird and frog sounds, mammal vocalizations are used on an as-needed basis, rather than predictably occurring at a particular time of day. While the acoustic structure of most mammalian vocalizations is considered to be relatively inflexible, evidence suggests that ground squirrels can shift the acoustic emphasis of their calls when challenged with a noisy environment (Rabin et al. 2003). California ground squirrels (*Spermophilus beecheyi*) in a noisier environment close to roads emphasized higher harmonics of their anti-predator calls than they did at less noisy sites (Rabin et al. 2003). Tree squirrels may adjust the rate of occurrence of their calling; eastern gray squirrels in relatively urban environments vocalized more often than did those in relatively rural areas when presented with playbacks of alarm calls via a robotic squirrel model (Partan et al. 2010). Because anti-predator calls cannot be adjusted for time of day as can the advertisement calls of frogs or birds, squirrels may increase the rate of calls or adjust the acoustic emphasis of the calls to transmit effective anti-predator signals in noisy urban areas.

Most of the literature on how animal communication systems respond to urbanization (discussed above and summarized in Table 9.3) has addressed alterations in the acoustic sensory channel alone. However, many animals rely heavily on other signal channels such as visual or chemical, which may impose additional environmental constraints. As opposed to the acoustic channel, in which urban areas are undeniably noisier than rural ones (Warren et al. 2006b), it is unclear whether the urban environment increases or decreases visual or chemical noise and complexity. Visually, urban areas generally consist of flat surfaces, large built structures, manicured foliage, and the motion of people and vehicles. Chemically, there are myriad anthropogenic odors and chemical pollutants. Future research should address how visual and chemical noise in urban environments compares with that in natural areas where plant life has complex shape, color, and movement, there are few flat surfaces, and there are a plethora of natural odors.

The above discussion (and Table 9.3) generally focuses on a single sensory channel at a time; however, animals communicate with multiple sensory systems simultaneously (Darwin 1872; Partan and Marler 1999). The role of environmental change in the evolution of this multimodal communication is not well studied (Bro-Jørgensen 2010). It is possible that as noise increases in one sensory channel and makes signal transmission more difficult, animals will shift their emphasis to sending or perceiving signals in another channel (Brumm and Slabbekoorn 2005; Partan 2013), termed a “multimodal shift” (Partan et al. 2010). Multimodal shifts in signaling behavior between sensory channels have been documented less often than have unimodal shifts within a channel, and rarely in the field. Nonetheless, Rabin et al. (2006) found that California ground squirrels living near acoustically noisy wind turbines used more visual vigilance behavior than did those not living near the turbines, and Partan et al. (2010) found that urban gray squirrels showed more alarm behavior after being exposed to visual alarm behaviors of a simulated conspecific than rural squirrels did, although both urban and rural squirrels responded to simulated audio alarms.

9.7 From Proximate Behavior to Ultimate Outcomes

Across taxa, successful urban species can be described as having a characteristic behavioral repertoire, for they utilize novel anthropogenic resources such as food and shelter, tolerate humans, and often have unusually small and stable home ranges (Gliwicz et al. 1994; McKinney 2002; McCleery 2010). These traits tend to be present in animals that are generalists in terms of diet and habitat needs. In contrast, species that have specialized requirements for food or habitat do not adjust well when conditions change during urbanization and are often extirpated from urban areas (McKinney 2002; McCleery 2010).

9.7.1 Behavioral Flexibility

A key characteristic allowing the more generalist animals to thrive in urban areas is behavioral flexibility. Animals that have an predisposition based on their evolutionary history to modify what they eat, where they live, and how they interact with conspecifics, competitors, and predators appear to be better suited for the challenges of environmental change that accompany urbanization (Tuomainen and Candolin 2011; Kralj-Fišer and Schneider 2012; but see Evans et al. 2011). While it may at first seem beneficial for all species to be flexible in the face of change, the ability to remain plastic can come with fitness costs for animals that live in a stable environment (Auld et al. 2010). For example, the maintenance of extensive brain tissue needed for behaviorally flexible traits such as learning can come at a cost of energetic investment in other life history traits that affect reproduction and survival (Mery and Kawecki 2003; Snell-Rood et al. 2011).

Individuals within a species have various degrees of flexibility in their behavior, or “behavioral reaction norms” (Dingemanse et al. 2010; Tuomainen and Candolin 2011) depending on how costs and benefits of flexibility affect fitness in a particular environment. Factors that can influence behavioral flexibility of a species were outlined by Tuomainen and Candolin (2011) who describe how behavioral reaction norms are shaped by evolutionary history, and then how physiological and other proximate factors influence how individuals can use their reaction norms to adjust to environmental change. Individuals from a population that evolved in variable environments are more likely to have a wider range for their behavioral reaction norm than individuals from a population that evolved in more stable environments (Tuomainen and Candolin 2011).

Many of the species that have successfully adjusted to urbanization or become synanthropes are considered to be edge species in rural environments (McKinney 2002; McCleery 2010). Edge species live in transitional areas between different types of habitat, which inherently are highly heterogeneous. The flexibility that edge species need to live in heterogeneous habitats contributes to their success in landscapes altered by humans; they may be able to exploit new resources more easily than other species that have evolved in more homogeneous environments.

In addition to understanding how flexibility expressed through behavioral reaction norms contributes to the success of urban animals, there is also increasing interest in understanding the relationship between flexibility and temperament. Individuals often respond in a consistent manner across contexts and can be considered to have temperaments, or “behavioral syndromes,” such as aggressive, explorative, or bold (for review, see Sih et al. 2004). While behavioral syndromes in urban environments is a new area of study, results suggest that more bold and less neophobic individuals may be better competitors in urban habitats (Short and Petren 2008; Sih et al. 2011), although this may be less clear-cut at the population level (Kralj-Fišer and Schneider 2012). In addition, the flexibility to be bold only when it is advantageous to do so is adaptive; an individual that is invariably bold across contexts may not fare as well as one that can match its behavior with current contextual challenges (Dingemanse et al. 2010). How temperament factors into decision making and flexibility in behavioral response is thus an area in need of further investigation, especially in relation to animals adjusting to urbanization.

9.7.2 Evolutionary Implications

Individuals can alter their behavioral response to environmental change within the limits of their genetically based behavioral reaction norms; alterations in behavioral response can come about through learning at the individual level or potentially through changes to the range of the reaction norm over generations (Sih et al. 2011; Tuomainen and Candolin 2011). As animals begin to adjust to urban environments, there is the potential that their change in behavior can ultimately result in a genetic change across the population, although Gienapp et al. (2008) suggested that many of the trait changes associated with a related rapid environmental shift, climate change, are likely due to phenotypic plasticity rather than genetic adaptation. However, the optimal suite of behaviors beneficial in the urban environment may be different from those beneficial in an animal's native environment. As animals adapt to city life, it is possible that directional selection towards a new optimum suite of behaviors is taking place within the population (Tuomainen and Candolin 2011). Additionally, new characteristics may be selected for or other traits may experience relaxed selection in new environments. For example, the primary selective pressures on Key deer (*Odocoileus virginianus clavium*), the availability of fresh water and food on islands, were largely removed by urbanization, allowing larger group sizes, greater densities, and decreased ranges (Peterson et al. 2005).

Any of the behavioral differences between urban and rural animals described in this chapter could potentially drive the evolution of genetic change in urban populations. To demonstrate that rapid evolution has occurred in urbanizing environments, however, genetic change must be documented (Gienapp et al. 2008); without genetic data, the changes cannot be distinguished from plastic phenotypic change. For example, rapid evolution has been claimed in urban bird behavior (reviewed in Diamond 1986), but few of the studies actually examined genetic data. Learning, plasticity, evolutionary history, and contemporary evolution interact in complex ways as animals strive to survive rapid environmental changes associated with urbanization (Sih et al. 2011). We are in pressing need of more work now to understand these interactions.

Conclusion

Animals faced with the challenges of urbanization must shift from a behavioral repertoire that was sufficient in their native environment to one that is closer to a new optimum for an urban environment. Some species and individuals can make this adjustment smoothly, by utilizing anthropogenic resources or by avoiding human activity, while others cannot or are outcompeted by newly introduced species. In this chapter, we reviewed behavioral changes in the use of space and structures, foraging, anti-predator behavior, social behavior, and communication, when animals live in urban areas. We also discussed the roles of behavioral flexibility and genetic evolution in adjusting to change. Understanding the effects of urbanization on animal behavior is important for conservation and management of urban species, and for improving human-animal coexistence. It will allow us to improve urban planning and the design of parks and other green spaces so that human-animal interactions are facilitated and biodiversity losses are mitigated.

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Chapter 10
Infectious Disease and Contaminants in Urban Wildlife: Unseen and Often Overlooked Threats

Seth P. D. Riley, Laurel E. K. Serieys and Joanne G. Moriarty

Keywords Disease · Spillover · Emerging infectious disease · Host–pathogen ecology contaminants · Rodenticide · Insecticide · Endocrine disruptors · Synergistic effect

10.1 Introduction

Urbanization drastically reduces and fragments wildlife habitat, but it can also generate less obvious effects that influence wildlife populations and communities. Two important influences that are often imperceptible without intense scientific investigation are infectious disease and contaminants. In its broadest sense, disease ("dis" and "ease," or the lack of ease) includes the effects of contaminants, but here we will use the term disease to refer to infectious diseases caused by living organisms, including microparasites, (e.g., bacteria and viruses) and macroparasites (e.g., arthropods and worms). Disease and contaminant effects can be cryptic, unless there are large, conspicuous die-offs. Furthermore, diseased animals are hard to find (unless radio-tracked), and even if they are found, lack of clear symptoms, lack of interest in cause of death, or lack of resources make diagnosis difficult.

The effects of disease and contaminants differ in important ways that have implications for understanding and mitigating the influence of urbanization on wildlife. While infectious diseases are caused by live pathogens, contaminants are generally manufactured chemicals introduced into the environment by humans. Infectious disease is an important factor in the evolution of all living species, often

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S. P. D. Riley (✉) · J. G. Moriarty
Santa Monica Mountains National Recreation Area, National Park Service,
401 W. Hillcrest Dr., 91360 Thousand Oaks, CA, USA
e-mail: seth_riley@nps.gov

L. E. K. Serieys
Dept. of Ecology and Evolutionary Biology, University of California,
Los Angeles, 621 Charles E. Young Drive, South, Los Angeles, CA, USA

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playing a strong role in natural selection. In turn, animals have shaped the evolution of their pathogens in a co-evolutionary arms race, where the host responds to the pathogen by developing immune responses and the pathogen responds to these defenses. This co-evolution often leads to a reduction in the virulence (or lethality) of the pathogen over time. Although human activities such as urbanization may introduce new parasites, the host's immune system may still have some experience with that or a similar infectious agent.

A contaminant is a substance normally absent in the environment that adversely effects living organisms when present in sufficient concentrations. With contaminants, unlike diseases, animals generally have no experience with the agent (e.g., for human made chemical compounds) or with the amounts encountered (e.g., for heavy metals), so immune responses are generally not possible, nor can they easily evolve. There are naturally produced chemicals (toxins), such as allelopathic compounds in plants or venom produced by animals, that animals have developed responses to over evolutionary time. Nonetheless, the bewildering array of chemical compounds designed and often mass-produced by humans have only been encountered by wild animals in their environments since the Industrial Revolution, leaving relatively little time to evolve adaptations. Although some insects have evolved resistance to pesticides, vertebrates have much smaller population sizes and longer generation times, making this more difficult. Moreover, humans continually produce new compounds to counter plant or animal adaptations. For example, when rats became resistant to the anticoagulant warfarin, more potent anticoagulant rodenticides were developed (see Sect. 10.3.4.1). Finally, because contaminants are produced through human activities, they are present at high concentrations in many urban systems.

As we consider how urbanization influences disease and contaminants in wildlife, a key factor will be whether landscape changes and human activity associated with urbanization affect the likelihood of exposure. With contaminants, one factor is whether the exposure is primary, where the animal acquires it directly from the environment, or secondary, where one animal is initially exposed and then a second animal acquires it from the first, as seen through predation. Similarly, diseases are both directly transmitted from an infected animal to an uninfected animal and indirectly transmitted through an intermediate host or the environment.

Wildlife ecologists increasingly recognize the importance of disease and contaminant issues for wild animal populations, and the research base on the topic is growing. In this chapter, we address how urbanization affects disease and contaminants in wildlife. We discuss what is known about disease and contaminants in urban landscapes, how pathogens and contaminants may interact, and the potential implications for wildlife in urban areas. In many cases, there may be limited or no information directly from cities, so we hypothesize, based on the literature from nonurban landscapes and our experience, about the potential implications for urban wildlife. Finally we discuss conservation and management policy relative to disease and contaminant issues.

10.2 Disease in Urban Wildlife

10.2.1 Transmission of Disease

The emergence or persistence of a pathogen relies on its ability to move from one susceptible host to the next, so the mode of pathogen transmission is an important consideration in understanding the impacts that urban areas have on disease prevalence. Direct transmission pathways are especially common within species and can occur through mating, intraspecific strife (fighting), interactions between parents and offspring, and casual social contact between members of a social group. Direct transmission pathways between species include predator–prey (including consumption) and aggressive interactions as well as more passive contact such as the use of the same feeding or watering sites. Alternatively, disease can be spread indirectly, by exposure to saliva, feces, or other fomites (contaminated objects) in the environment, or through vectors or intermediate hosts, which can be a part of the elaborate pathogen life cycles. Host ecology affects disease prevalence by influencing the opportunity for pathogen transmission and the susceptibility of hosts (Altizer et al. 2006), so that changes in host ecology that affect contact between individuals can have substantial effects on transmission rates and thereby prevalence of the disease. This is often the case for "emerging infectious diseases" (EIDs), diseases that are changing in prevalence or host range or that are evolving new variants (Daszak et al. 2000, 2001). Urbanization, through the elimination, fragmentation, and alteration of habitat, has a huge effect on the ecology of host animal populations, and scientists are just beginning to elucidate its significance for disease dynamics in wildlife.

Many diseases are dependent on direct contact because the pathogen does not persist for long in the environment (e.g., mange caused by mites), so the changes in contact rates may have profound effects on the opportunity for transmission (Muller et al. 1989). Consequently, changes in behavior or demography in urban wildlife populations that increase the rate of direct contact may lead to increased prevalence and impacts of disease. Some diseases, including the rapidly emerging chronic wasting disease that affects cervids (Farnsworth et al. 2005), are more complex because they spread through both direct and indirect transmission (Miller et al. 2004), and both forms of transmission may be affected by urbanization.

Trace amounts of feces or fluids, such as nasal and respiratory exudate, saliva, ocular secretions, and urine, all have the potential to spread diseases indirectly when distributed on fomites in the environment. For example, canine parvovirus (CPV) and canine distemper virus (CDV) both spread easily through this type of indirect contact. Both of these EIDs (Deem et al. 2000; Steinel et al. 2001) are of concern for carnivores, and CDV in particular can have significant population consequences for some species, including endangered African wild dogs near towns (Table 10.1; Alexander and Appel 1994) and raccoons (*Procyon lotor*) in cities (e.g., Roscoe 1993). CPV is primarily transmitted through a fecal–oral route (i.e., through the ingestion of contaminated fecal partials) and is extremely stable in the

Table 10.1 Studies that compared disease in wildlife between urban and non-urban areas. Studies are organized by the proposed mechanism for differences in disease prevalence or effects in the urban landscape

Pathogen	**Host**	**Location**	**Urban shift or effect**	**Mode of transmission**	**Potential urban factors**	**Study**
Proximity to humans and their animals (six cases)						
Canine distemper virus	South American Foxes (*Lycalopex griseus*, *Lycalopex culpaeus*)	Chile	Both foxes and unvaccinated domestic dogs had greater seroprevelance with increasing proximity to urban settlements	Primarly ingestion of aerosoled respiratory exudate, also ingestion of bodily fluids and fecal matter	Disease reservior and spillover, host density	Acosta-Jamett et al. 2011
Canine distemper virus	African wild dogs (*Lycaon pictus*)	Kenya	Disappearence of local African wild dogs concurrent with epizootic of CDV in domestic dogs in towns	Primarly ingestion of aerosoled respiratory exudate, also ingestion of bodily fluids and fecal matter	Disease reservior and spillover, host density	Alexander and Appel 1994
Canine parvovirus	Gray foxes (*Urocyon cinereoargenteus)*	California, USA	Greater seroprevalence in urban vs. rural areas	Fecal–oral, exposure to infected carnivore feces	Spillover—from domestic dogs	Riley et al. 2004
Toxoplasma gondii	Woodchucks (*Marmota monax*)	Illinois, USA	Increased seroprevalence with greater urban association	Ingestion of infected meat, exposure to infected felid feces	Spillover—increased density of domestic felids, increased interaction of wild felids	Lehrer et al. 2010
Toxoplasma gondii	Southern sea otters (*Enhydra lutris nereis*)	California, USA	Greater seroprevalence at areas of maximum freshwater input (largely from untreated urban runoff)	Ingestion of infected meat, exposure to infected felid feces	Pathogen pollution spillover	Miller et al. 2004

Table 10.1 (continued)

Pathogen	**Host**	**Location**	**Urban shift or effect**	**Mode of transmission**	**Potential urban factors**	**Study**
West Nile virus	Birds	Georgia, USA	Positive correlation between seropositive birds and urban/suburban land use	Vector-mosquito (Culex spp.)	Possible spillover—fewer overall positives, but stronger association with urbanization in first year	Gibbs et al. 2006
High host density (five cases)						
Canid tapeworm (*Echinococcus multilocularis*)	Red foxes (*Vulpes vulpes*)	Western Europe	Most prevalent in urban periphery and rural/urban intersection	Ingestion of intermediate host (small mammals), or fecal–oral from definitive host (canids)	Resource availability/high host density combined with highest intermediate host density	Deplazes et al. 2004
Mycoplasmal conjunctivitis (*Mycoplasma gallisepticum*)	House finch (*Carpodacus mexicanus*)	Eastern North American	Novel strain of bacterium typically found in poultry emerged in house finches, finches are highly associated with urbanization on East Coast	Infected eggs	Resource availability/high host density, spillover, resource clumping	Dhondt et al. 2005
Raccoon roundworm (*Baylisascaris procyonis*)	Allegheny woodrat (*Neotoma magister*) (intermediate host)	New York and New Jersey, USA	Persistence of reintroduced wood rat populations was negatively correlated with density of raccoon latrines (raccoons, the definitive host, are known to be more dense in urban areas)	Fecal–oral from definitive host to intermediate and definitive host, ingestion of intermediate host by definitive host	Resource availability/high definitive host density	LoGiudice et al. 2003

Table 10.1 (continued)

Pathogen	**Host**	**Location**	**Urban shift or effect**	**Mode of transmission**	**Potential urban factors**	**Study**
Unknown	Raccoons (*Procyon lotor*)	Illinois, USA	High mortality in urban vs. rural animals in 1 of 3 study years caused by unknown disease	Unknown	Resource availability/ high host density- urban and suburban densities were significantly higher than rural, spillover, resource clumping	Prange et al. 2003
Sarcoptic mange	Red foxes (*Vulpes vulpes*)	Illinois, USA	Higher mange prevalence in urban areas and urban/rural edge areas	Direct contact	High host density in urban and urban/ rural edge areas, potentially from reduced mortality/ exclusion by coyotes	Gosselink et al. 2007
Reduced biodiversity (five cases)						
Dixenous helminths	Red foxes (*Vulpes vulpes*)	Switzerland	Higher prevalence in rural vs. urban areas	Ingestion of intermediate host (small mammals)	Low occurrence of intermediate host, anthropogenic food available	Reperant et al. 2007
Intestinal helminth parasites	Red foxes (*Vulpes vulpes*)	United Kingdom	Different helminth assemblages in urban and rural foxes	Ingestion of intermediate host (small mammals)	Reduced biodiversity and alternate resource availability producing differences in diet	Richards et al. 1885

Table 10.1 (continued)

Pathogen	Host	Location	Urban shift or effect	Mode of transmission	Potential urban factors	Study
Lyme disease (*Borrelia burgdorferi*)	White-footed mouse (*Peromyscus leucopus*)	New York, USA	Higher mouse density in smaller habitat fragments, and greater density of infected ticks	Vector—blacklegged tick (*Ixodes scapularis*)	Reverse dilution effect—white-footed mouse is the most competent host and persists in degraded habitats where other local species disappear	Allen et al. 2003
West Nile virus	Passerines (Passeriformes)	Louisiana, USA	Greater density of infected mosquitos with lower nonpasserine diversity (correlation of diversity with urbanization not investigated)	Vector—mosquito (Culex spp.)	Reverse dilution effect	Ezenwa et al. 2006
West Nile virus	Passerines, especially robins	All of North America	Higher prevalence in human-dominated areas, both urban and agricultural	Vector—mosquito (Culex spp.)	Reduced bird diversity in urban and agricultural areas/ Also resource availability high vector and host density in human-associated areas	Kilpatrick et al. 2011

Table 10.1 (continued)

Pathogen	Host	Location	Urban shift or effect	Mode of transmission	Potential urban factors	Study
Alteration of landscape structure and uniformity—resource clumping (three cases)						
Endoparasitic nematodes	Raccoons (*Procyon lotor*)	New York, USA	Greater endoparasite infestations with higher contact rates (correlation of contact rates with urbanization not investigated)	Ingestion, fecal–oral	Resource clumping	Wright and Gompper 2005
Chronic wasting disease	Mule deer (*Odocoileus hemionus*)	Colorado, USA	Urban deer two times more likely to test positive than nonurban deer	Unknown, thought to be direct and indirect, very persistent in the environment	Resource clumping, Resource availability/high host density, refuge from predation for sick animals, possibly spillover from captive animals has moved prion around	Farnsworth et al. 2005
Squirrel parapoxvirus	Red squirrels (*Sciurus vulgaris*)	United Kingdom	Unknown (originally introduced with the intentional introduction of gray squirrels, an urban adapted species)	Direct contact with open lesions, potentially mosquito vectors and fomites)	Resource clumping	Tompkins et al. 2003
More dynamic landscapes—increased stress						
Nonglucocorticiod acute stress response	European blackbird (*Turdus merula*)	Germany	Smaller corticosterone response to acute stress in winter and spring, but not first fall	Nonmediated by acute stress	More dynamic landscapes, more novel stressor	Partecke et al. 2006

Table 10.1 (continued)

Pathogen	**Host**	**Location**	**Urban shift or effect**	**Mode of transmission**	**Potential urban factors**	**Study**
Contamination and Pollution						
Mange (Notoedres cati)	Bobcats (*Lynx rufus*)	California, USA	Substantial urban mortality, no observed mortality in more natural area	Direct contact	Contamination (association with exposure to anticoagulant rodenticides), high host density	Riley et al. 2007; Moriarty unpublished data
Nonstress biomarkers	Great tit (*Parus Major*)	Sweden	Greater levels of oxidized glutathione, an oxidative stress biomarker, were detected in urban vs. rural environments	Nonoxidative stress has been shown to be caused by ingested or inhaled pollutants, primarily traffic-generated air pollution	Pollution	Isaksson et al. 2005

environment. Feces are broadly distributed throughout the natural environment and are also regularly deposited by domestic animals such as cats and dogs. Indirectly transmitted diseases may be of particular concern as zoonoses (disease transmitted from animals to humans) because of the higher likelihood of human exposure to fecal or bodily fluid residues in the environment, particularly in urban settings. For example, *Toxoplasma gondii*, which causes toxoplasmosis, can cause serious disease in humans and can be spread indirectly from definitive hosts, specifically felids (Luft 1989).

Some pathogens with more complicated life cycles have vectors or intermediate hosts that transmit pathogens indirectly between definitive hosts. Pathogens spread by vectors, including those spread by ticks, mosquitos, and fleas, can be of particular concern for human health, because people are commonly exposed to these invertebrate vectors even when they might not be exposed to the definitive hosts. West Nile virus, a primarily avian pathogen spread by mosquitos, has caused widespread disease in humans throughout the world and may be more prevalent in urban areas (Gibbs et al. 2006). Lyme disease, spread by ticks, and plague, spread by fleas, are other zoonotic diseases that may present a risk to human health in urban regions.

10.2.2 Factors Influencing Wildlife Disease in Urban Areas

Changes in host–pathogen ecology may cause important changes in pathogen emergence or pathogen prevalence (Daszak et al. 2001; Woolhouse 2002). Many characteristics of urbanized areas create ecological changes affecting the dynamics of wildlife diseases, including environmental or landscape disturbance, changes in resource availability, and changes in the diversity or distribution of the local flora or fauna. Further, in some cases, other types of disturbances (e.g., exposure to contaminants) may interact synergistically with pathogens to produce significant impacts for host populations.

Although there has been more attention paid in recent years to the issue of disease in urban wildlife (e.g., Bradley and Altizer 2007), the area of study is still a very young one. In a recent review of wildlife disease in human-modified landscapes, Brearley et al. (2013) analyzed 19 studies, at most 10 of which specifically addressed urban areas, and among 70 studies of bird parasites in urban areas, only 13 addressed spatial patterns and 6 evaluated population effects (Delgado-V and French 2012). For this chapter, we have attempted to find as many studies as we could that addressed disease in wildlife in both urban and nonurban landscapes (Table 10.1) and evaluated the most important ways in which urban environments have been found to affect wildlife disease dynamics.

10.2.2.1 Spillover and Spillback: Increased Disease Risk from Human-Associated Animals (Six Cases in Table 10.1)

In urban environments, the most obvious and perhaps the most important factor for infectious disease is the proximity of wildlife to humans, human-associated animals (HAAs; pets, livestock, and captive animals), and human-dependent wild animals (e.g., rats or pigeons). Human activities can move pathogens quickly and across large distances, introducing them to naïve populations. Human travel, the animal trade, including pets, zoos, and laboratory animals, and the animal food trade (Karesh et al. 2007) all have the potential to transport pathogens to new places or to create “pathogen pollution” (Cunningham et al. 2003). Daszak et al. (2003) implicated pathogen pollution in the introduction of chytridiomycosis to naïve amphibian species around the world, which has had dramatic consequences for many species, including extinctions. Urban areas can be centers of introductions of exotic species. Introductions can occur intentionally for aesthetics or biocontrol or unintentionally from accidental releases, but any introduction has the potential to introduce novel pathogens to the urban environment. For example, gray squirrels (*Sciurus carolinensis*) were intentionally released into human dominated areas of the UK in the early 1900s and are thought to have introduced and maintained squirrel parapoxvirus, a disease potentially responsible for the dramatic decline of the native red squirrel (*Sciurus vulgaris*) population (Tompkins et al. 2003).

Dense populations of HAAs, especially domestic cats (*Felis catus*) and dogs (*Canis familiaris*), facilitate pathogen “spillover” of already established diseases to wildlife. The flow of disease from HAAs to wildlife can infect naïve populations, and HAAs can function as disease reservoirs allowing a continuous source for reemergence of diseases in wildlife. Riley et al. (2004) found that gray foxes (*Urocyon cinereoargenteus*) in northern California had greater seroprevalence of CPV in an urban area than in a nearby rural area, and the only seropositive animals at the rural site lived near the largest town. Similarly, “spillback,” the flow of disease to HAAs from wildlife, can have potentially important implications for wildlife and human health (Daszak et al. 2000). Spillback can help to perpetuate HAA populations as disease reservoirs as well as produce disease epizootics in livestock or other HAAs, an often politically charged event. Acosta-Jamett et al. (2011) showed that both foxes and unvaccinated domestic dogs in the Coquimbo region of Chile had greater seroprevalance for CDV with increasing proximity to urban settlements. Greater seroprevalence indicates higher levels of disease exposure, and the urban domestic dog population was suspected of being the source for an epizootic in wild foxes. Carnivore and ungulate diseases may be of particular concern in terms of spillover and spillback, as domestic dogs and cats, as well as some livestock, are often allowed to roam freely in areas commonly visited by wildlife. Places where pets are fed outdoors, and especially feeding sites in natural or park areas (such as for feral domestic cats), may attract many HAAs as well as wildlife, and pet food bowls, because of their inorganic and often porous composition, can harbor pathogens particularly well.

10.2.2.2 High Host Densities Increase Host Contact Rates (Four Cases in Table 10.1)

The increased densities that are reached by some urban wildlife populations (Chap. 8) can alter host behavior and ultimately increase contact rates (both direct and indirect) between infected and susceptible individuals. Increased host density and therefore contact rates have in turn been associated with greater disease prevalence (Table 10.1; e.g., in raccoons; Prange et al. 2003). As with spillover and spillback, increased density may be of particular concern within taxonomic groups that are represented in the HAA populations, such as carnivores and ungulates. Additionally, the combination of domestic and wildlife populations increases the overall effective host population size and density for the disease, both of which can increase the likelihood and severity of outbreaks. As with spillover and spillback, high host densities can also contribute to zoonotic disease. The canid tapeworm (*Echinococcus multilocularis*) and raccoon roundworm (*Baylisascaris procyonis*) both can cause serious disease in humans (Scheuhammer Schicker 1976; Sorvillo et al. 2002), and both have shown the potential for increased prevalence in urban landscapes (Table 10.1; LoGiudice et al. 2003; DePlazes et al. 2004) where red fox (*Vulpes vulpes*) and raccoon densities can be high (Hadidian et al. 2010; Soulsbury et al. 2010).

10.2.2.3 Resource Clumping Increases Host Contact Rates (Three Cases in Table 10.1)

Fragmentation in urban areas results in small, isolated natural areas within heterogeneous landscapes where resources are likely to be more aggregated than in natural landscapes. This clumping of resources may lead to greater direct and indirect contact rates, both within and among species, which in turn can lead to greater rates of disease transmission. In an interesting example, Wright and Gompper (2005) found that raccoons had greater contact rates, greater prevalence of endoparasite infestations, and greater evenness and diversity of endoparasite prevalence when their resources were more clumped, suggesting both that transmission was limiting for these parasites and that increased contact brought on by changes in resource dispersion could affect transmission rates.

10.2.2.4 Reduced Animal Diversity Alters Host–Pathogen Contact Rates (Four Cases in Table 10.1)

The intense alteration of urban areas leads to fewer animal species, and these reductions in host species richness and evenness can in turn affect the diversity and abundance of pathogens by affecting the rate of contact between potential hosts. Specifically, in more species rich areas, the contact rate between competent hosts is diluted by the presence of less competent host species (Schmidt and Ostfeld 2001;

Keesing et al. 2006). Reduced biodiversity may alter host–pathogen ecology by increasing the relative density of some species, as competing species are reduced or eliminated, and therefore reversing this "dilution effect." Subsequently, the rate of transmission of pathogens between competent hosts may increase. This effect may be relevant for many types of pathogens, but it is particularly relevant for vector-borne diseases where the vectors typically feed on multiple hosts (Keesing et al. 2006). A model produced using actual tick burdens and nymphal infection rates of lyme disease from various vertebrate species in Duchess County New York showed that greater species diversity was negatively correlated with nymphal infection prevalence (LoGiudice et al. 2003).

10.2.2.5 Stress, Microclimate Shift, and Interactions with Contaminants Affect Disease Susceptibility

Construction activities, infrastructure or landscape maintenance, and seasonal and diurnal variation in human activity make urban areas highly dynamic, potentially inducing additional stress for wildlife. The link between stress and susceptibility to disease is established in humans (Godbout and Glaser 2006) and wildlife (Owen et al. 2012). However, there is evidence that some wildlife may be able to adapt to the increased levels of stress in urban areas. The corticosterone response of common blackbirds (*Turdus merula*) to acute stress in a laboratory setting was reduced in birds collected from urban areas relative to their rural counterparts raised in the same environment (Partecke et al. 2006). A decreased stress response could reduce disease susceptibility for animals faced with greater disturbance levels in urban landscapes.

The microclimate shifts that often occur with urbanization as cities become "heat islands" (Chap. 4) may also alter disease dynamics in urban wildlife. Between 1997 and 2000 in Phoenix, AZ, USA, the thermal window for arthropod reproduction was 41 days per year longer in the warmest, highly urbanized area than it was in the cooler rural area nearby (Baker et al. 2002). This increased reproductive window could facilitate the greater abundance of vector species (e.g., mosquitos or ticks). Further, mild climates combined with high resource availability in urban areas may prolong the survival of diseased animals, increasing their opportunities of transmitting the disease to naïve individuals.

Wildlife populations are exposed to contaminants in many different environments, from agricultural to urban landscapes. However, as discussed in Sect. 10.3 (below), more wildlife species may be exposed to a greater array of contaminants in urbanized landscapes than anywhere else. Along with their direct effects, these contaminants can interact with pathogens or hosts, increasing susceptibility to disease (see Sect. 10.3.3.3).

10.2.3 Impacts of Disease in Urban Areas

We have described multiple pathways by which urbanization may affect or alter disease dynamics in urban wildlife, but what are the impacts of disease on wildlife in cities? There are two reasons why disease impacts may be more common in wildlife populations in urban areas. The high densities of some wildlife species in urban landscapes mean that host population size and density will be larger (Sect. 10.2.2.2). First, many diseases are strongly density-dependent, so these larger and denser populations will likely lead to greater disease incidence and more epizootics. Even for populations that are relatively large, disease epizootics, especially of diseases that are highly virulent, can be devastating. There are many cases of disease epizootics in urban areas having major impacts on populations that were initially dense. For example, raccoons in Rock Creek Park in Washington, D.C. reached some of the highest densities recorded for the species, but the population was greatly reduced by the mid-Atlantic rabies epizootic (Riley et al. 1998), and urban raccoon populations have suffered from disease (CDV) in other areas as well (Hoff et al. 1974; Roscoe 1993). Similarly, the red fox population in Bristol, England, which had also reached high densities, was reduced by sarcoptic mange (Soulsbury et al. 2007). In a study spanning urban, rural, and edge areas in Illinois, high density red fox populations in urban areas were affected most by sarcoptic mange, while foxes in rural areas, where coyote (*Canis latrans*) predation predominated and fox density was lower, had the lowest mange incidence (Gosselink et al. 2007; Crooks et al. 2010). Epizootics like sarcoptic mange may be more common, and potentially more damaging, in high-density urban wildlife populations.

The second reason for increased disease impacts in urban areas is that proximity of urban wildlife to HAAs may expose wildlife to pathogens more often. This may be of particular concern for species that are sensitive to certain diseases or for species of conservation concern like the endangered Channel Island fox (*Urocyon littoralis*, Clifford et al. 2006). On the island of Catalina, the fox population was devastated in 1999–2000, likely by an epizootic of canine distemper (Timm et al. 2009). The CDV exposure was thought to have come from stowaway raccoons from boats or from domestic dogs. Similarly, the endangered San Joaquin kit fox (*Vulpes macrotis mutica*) in urban areas in the Central Valley of California faces exposure to rabies and other felid diseases through regular interactions at den sites and cat-feeding stations (Harrison et al. 2011). Intense habitat fragmentation of urban landscapes means small populations of common species are also subject to local extirpation from factors such as disease. In the case of notoedric mange in bobcats in southern California, one habitat fragment completely surrounded by roads and development had a high bobcat density of at least six radio-collared bobcats using a 3.0 km^2 habitat fragment before the epizootic. Five bobcats died of mange in 2002 and 2003, and for a number of years afterwards, regular trapping, scat surveys, and radio-tracking of animals in nearby areas showed little or no use of this fragment other than by occasional males. However, more recently, we have captured and radio-collared multiple bobcats, including reproductive females, in this fragment (Fig. 10.1), indicating some recovery. Bobcats are highly mobile, and

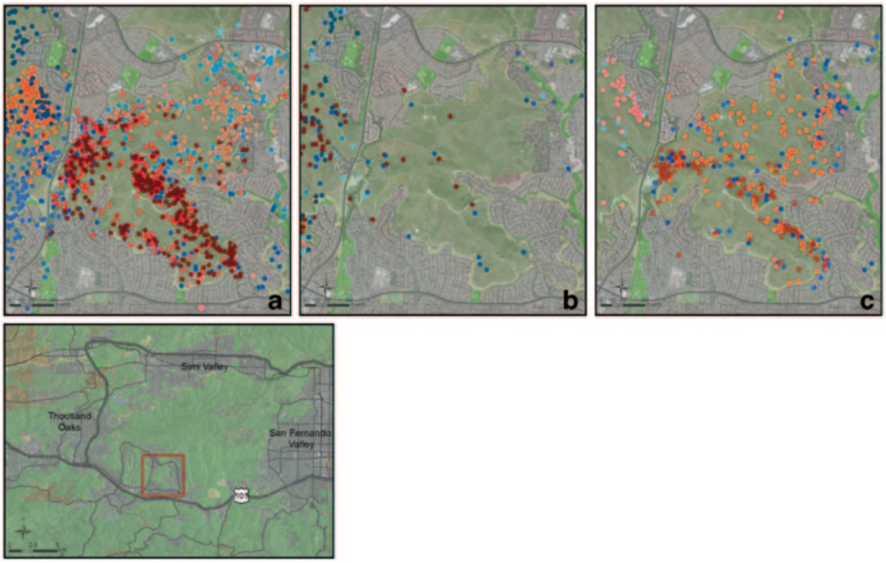

Fig. 10.1 An urban habitat fragment in Ventura County, California, with telemetry (VHF and GPS) locations for male (*blue*) and female (*red* and *orange*) bobcats for the periods 2001–2002 (**a**), 2006–2007 (**b**), and 2011–2012 (**c**). The bobcat population in this area experienced an epizootic of the ectoparasitic disease notoedric mange beginning in late 2002, which was particularly intense in 2002–2005, and largely was over (in this area) by 2008

this habitat fragment is not far from larger core natural areas that could serve as a source of dispersing animals. But for more isolated subpopulations or less mobile species, subpopulations eliminated by disease in fragmented urban landscapes may be difficult to re-establish without human assistance.

10.3 Contaminants and Urban Wildlife

10.3.1 Chemical Contamination of Urban Ecosystems

The environmental catastrophe surrounding the use of dichlorodiphenyltrichloroethane (DDT) and its analogs raised public and governmental awareness of the environmental and societal costs associated with chemicals used against agricultural and public health pests (Carson 1962). Despite an increased awareness of the risks associated with environmental contaminants, worldwide use of pesticides has dramatically increased since the 1950s, with little understanding of how the majority of these pesticides impact the environment (Dich et al. 1997). Within the European Union alone, 100,000 pesticide chemicals were marketed as of 1998, with insufficient ecotoxicity data available for approximately 75 % of the 2000–3000 chemicals used most frequently (Gee 1998). The United States Environmental Protection

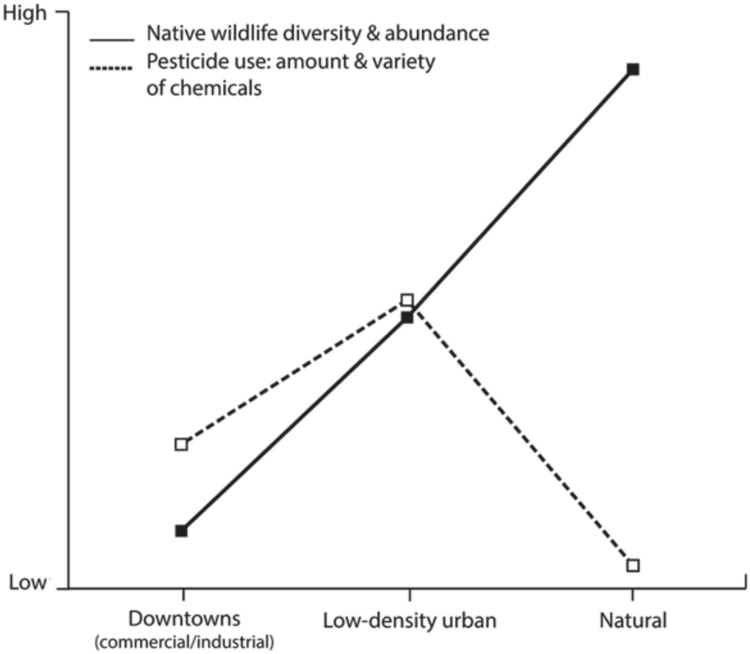

Fig. 10.2 Schematic of the relative amounts of pesticide use (insecticides, rodenticides, herbicides) and native wildlife diversity and abundance across the gradient from intensely urbanized downtown areas, through lower density urban areas, to natural open space. In certain low-density urban and suburban parts of the landscape, where native wildlife diversity and abundance may be relatively high, contaminant exposure and impacts may be particularly important

Agency (EPA) estimates that annually within the country, domestic users purchase 101 billion pounds of active ingredients for pesticides (Dalaker and Naifeh 1998). Although some of the most dangerous pesticides have been banned in developed nations, many of those pesticides are still produced and sold in developing countries (Mansour 2004). Contaminants that can affect wildlife include pesticides (insecticides, herbicides, rodenticides, avicides), industrial byproducts, pharmaceuticals, and fertilizers. The immense diversity of environmental contaminants generated through anthropogenic activities that can have serious, though often cryptic, impacts on wildlife populations makes examining the consequences of their use important.

Based on the distribution and abundance of native wildlife populations and contaminant use, wildlife species in some portions of the urban landscape may be particularly at risk from contaminants. In densely developed urban centers (downtown), while a few nonnative species may reach high densities (Chap. 7, e.g., rats (*Rattus* sp.), house mice (*Mus musculus*), pigeons (*Columba livia*), and house sparrows (*Passer domesticus*)) the diversity and density of native wildlife are likely to be low. In these urban centers, the lack of green space means the use of contaminants aimed at killing live organisms (insecticides, herbicides, rodenticides) is often low, and while there may be more exposure to industrial contaminants, these chemicals are not the greatest direct threat to wildlife. The mix of low native wildlife diversity and limited use of chemicals intended for killing live animals means relatively few wildlife toxicant problems are likely downtown.

However, many low density urban or suburban areas can have relatively diverse and sometimes abundant populations of native wildlife (Chap. 7). These same urban and suburban landscapes are likely to experience significant application of contaminants to control populations of unwanted plants and animals, specifically the rodenticides, insecticides, and herbicides that target living organisms and are likely to affect wildlife. The amount of use may not be as great or as widespread as in areas of intense agriculture, but it is likely much greater than that in urban centers, and certainly more much more than in natural areas (Fig. 10.2, dashed line).Therefore, high levels of pesticide use may coincide with relatively diverse and abundant wildlife communities in low-density urban and suburban parts of the landscape (Fig. 10.2). Accordingly, urban wildlife may be particularly exposed to contaminants and likely to suffer adverse effects from them. The great variety of contaminants, coupled with the lack of knowledge of their effects in natural systems, and in combination, present an added challenge. Finally, although contaminants may result in direct mortality or obvious reproductive or morphological effects, they may also interact with other stressors in the environment to affect individuals and populations in totally unforeseen ways (see Sect. 10.3 below).

10.3.2 Circumstances of Exposure

Chemical contamination of ecosystems can occur intentionally or unintentionally and can affect air, soil, and water, along with the plants and animals. Exposure of wildlife to chemical contaminants can have both direct consequences such as acute toxicity or sublethal physiological changes, and indirect consequences such as altered predator–prey dynamics or resource availability. Pesticides used to control invertebrate populations in Britain provide an example of indirect effects, where reduced invertebrate populations, the principal food source for gray partridges (*Perdix perdix*), led to decreased chick survival and thus decreased effective population size (Rands 1985). More recently, Mills and Semlitsch (2004) found the insecticide carbarmyl affected southern leopard frogs (*Rana sphenocola*) by altering the intensity of competition and predation and by changing the aquatic community structure. Both direct and indirect consequences should be considered when examining the role of pollutants on the population dynamics of urban wildlife. However, here we will largely focus on direct consequences of exposure to contaminants, because more is known about these effects and the impacts are generally more significant, at least in the short term.

Exposure to contaminants can be either primary or secondary. Primary poisoning involves direct exposure events (e.g., ingestion, inhalation, dermal contact), which frequently occur for insecticides, avicides, and rodenticides (Berny 2007). Secondary poisoning occurs when a predator or scavenger consumes poisoned or contaminated prey. Although secondary poisoning is usually associated with compounds with delayed action, such as anticoagulant rodenticides, it can also occur with toxicants with more rapid onset of effects, such as organophosphates and carbamate insecticides.

There are many specific ways for wildlife to be exposed in urban landscapes. The most direct way is through target or nontarget exposure to rodenticides or avicides directed at nuisance mammals or birds. Animals that prey or scavenge on these primarily exposed animals can then be secondarily exposed. Exposure to other contaminants may be more indirect. For example, although much of the use of insecticides and herbicides occurs in agricultural areas, they are regularly used in gardens, yards, and landscaped areas in urban areas. Terrestrial wildlife can then be exposed secondarily by eating contaminated insects. Many kinds of contaminants also end up in water bodies and wetlands, where aquatic wildlife species such as amphibians, aquatic reptiles, aquatic mammals, and waterbirds can be exposed. For all pesticides (rodenticides, herbicides, insecticides), exposure in wildlife may be more likely when applications are made that do not follow the label. For example, the common herbicide Roundup, or glyphosate, is not intended for use near water, but when used there it can have significant effects on amphibians (Relyea 2005).

10.3.3 Consequences of Exposure

Pollution of the environment by contaminants represents a major threat to wildlife and was identified as the third most important cause of endangerment in the the USA (Wilcove and Master 2005). The consequences of contaminant exposure vary among species and populations but may include direct mortalities and population declines (Ratcliffe 1970; Peakall et al. 1973; Lundholm 1997; Davidson et al. 2002; Riley et al. 2003), reproductive impairment (Vos et al. 2000; Hayes et al. 2010), and decreased immune competence and increased disease susceptibility or emergence (Vos et al. 2000; Kiesecker 2002; Ross 2002; Riley et al. 2007). Overall, the effects of contaminant exposure vary with the intensity and duration of exposure to the toxicant (Long et al. 1995). Divergent responses to pollutants by different species can complicate our ability to predict the effects of contaminant exposure on wildlife based on laboratory results (Thompson 1996). Further, it can be difficult to determine the mechanism of action of chemical contaminants in the field, where toxic effects can result from simultaneous exposure to multiple compounds, including multiple different types of chemicals (e.g., metals and organic pollutants; Vos et al. 2000).

Urban wildlife is vulnerable to all of the chemical effects that we, as humans, go to great lengths to avoid. For example, humans often do not drink, or even swim in, the water from the rivers and streams that run through urban areas. We stay out of the ocean when urban runoff has brought chemicals into and down streams after a storm. We have to breathe the air in our cities, but we stay inside and do not exercise on "bad air days." And, we import much of our food, and generally all of the fresh food (e.g., produce, meat, fish), from outside urban areas (Chap. 1). Yet, wildlife do not have these choices; they must breathe the air, live in or drink the water, and find food resources in these urban environments.

10.3.3.1 Direct Mortality

The most obvious way that contaminants affect wildlife in urban areas is by directly causing mortality, and essentially all groups of environmental contaminants have been linked to direct mortality in wildlife. DDT and its analogs were linked to mass mortalities in multiple wildlife species (Carson 1962) and more recently in dolphins along urban coastlines (Aguilar and Borrell 1994; Lahvis et al. 1995). Mendelssohn and Paz (1977) documented mass mortality of raptors and owls that fed on voles and birds poisoned by Azodrin, a chlorinated hydrocarbon. Approximately 400 birds of prey died within an 8-km^2 area during a 3-month period after the initial application of the pesticide. In southern California, secondary exposure to anticoagulant rodenticides was the second most important source of mortality for urban coyotes (Riley et al. 2003; Gehrt and Riley 2010).

10.3.3.2 Reproductive Impairment

Even moderate contamination can affect reproduction in wildlife populations (Scheuhammer 1987; Peakall 1992; Vos et al. 2000; Guillette 2000; Eeva et al. 2003). Contaminants that impact reproduction are found among nearly all types of chemicals used in urban areas, including anticoagulant rodenticides, insecticides, heavy metals, and industrial effluents. Primary effects of contaminants may include reduced female fertility and increased chance of miscarriage, reduction in sperm counts, altered gonad development, and hormonal or endocrine disruption (Vos et al. 2000; Munday and Thompson 2003; Choi et al. 2004; Robinson et al. 2005). Laboratory experiments showed that lead exposure affected seminal vesicles and epididymes in cotton rats (*Sigmodon hispidus*; McMurry et al. 1995). Hormonal disturbances in wildlife caused by endocrine disruptors include sex changes in riverine fish and marine snails, reproductive failure in birds, and abnormalities in the reproductive organs of American alligators (*Alligator mississippiensis*; reviewed in Vos et al. 2000). Anticoagulant rodenticides, the most commonly used rodenticides worldwide, are associated with increased chance of miscarriage, in utero toxicity, and decreased sperm counts (Munday and Thompson 2003; Robinson et al. 2005). The breeding success for two bird species, blue tit (*Cyanistes caeruleus*) and great tit (*Parus major*), was reduced as a result of exposure to heavy metal pollution from a copper smelter (Eeva et al. 2003, 2009).With the varied and sometimes intensive use of contaminants in urban landscapes, reproductive impacts may be much more common than we know.

10.3.3.3 Immunological Impairment and Increased Susceptibility to Other Stressors

The vertebrate immune system is affected by the toxicological activity of many xenobiotics (chemicals that are foreign to living organisms), with examples from both the lab and wildlife in the field. Endocrine disruptors, such as DDT, methyl mercury, and polychlorinated biphenyls (PCB), are the chemicals most often associated with altered immune response among humans and wildlife. The immune consequences of exposure to these contaminants can vary according to dose and length of exposure, but they can include reduction in antibody production, loss of proliferation of T-lymphocytes when exposed to mitogens (a mitogen is an agent that induces mitosis, especially lymphocyte proliferation), and reduced cytotoxic responses of natural killer cells. DDT has been shown to have immunotoxic effects in both mammals and birds in the lab, leading to a reduction in antibody response and increased susceptibility to infection (Vos and Moore 1977). Laboratory investigations have also found that numerous common pollutants such as PCBs, polybrominated biphenyls, and 2,3,7,8-tetrachlorodibenzo-p-dioxin have effects on both humoral and cell-mediated immunity in mammalian and avian species (Faith et al. 1980; Exon et al. 1985). In wild bottlenose dolphin (*Tursiops truncatus*) populations, peripheral blood lymphocytes had decreased proliferative responses to T-cell mitogens correlated with high levels of DDT and PCBs (Lahvis et al. 1995). Facemire et al. (1995) suggested that endangered Florida panthers (*Puma concolor coryi*) suffer immunosuppression as a result of exposure to endocrine disruptors such as mercury, dichlorodiphenyldichloroethylene (DDE), and PCBs. Not surprisingly, given the effects on the immune system, contaminant exposure has been linked to increased disease susceptibility and incidence in fish (Vethaak et al. 1996; Arkoosh et al. 1998, 2001), mammals (Ross 2002; Aguilar and Borrell 1994; Riley et al. 2007), and amphibians (Rohr et al. 2008).

There can be a synergistic effect between anthropogenic stressors, such as contaminant exposure, and natural stressors, including disease, predation, or competition, in urban wildlife populations (Sih et al. 2004). Experiments with amphibians suggest that combined anthropogenic and natural stressors, such as pesticides and predation, have a greater influence on survival, growth, and population persistence than either factor alone (Kiesecker 2002; Relyea 2003). Food stress (dietary protein insufficiency) and mild physical or emotional stress all increased immunotoxicity as a consequence of contaminant exposure in laboratory mice (Banerjee et al. 1997; Banerjee 1999). If a wider variety of species are exposed to a wider variety of different contaminants (which may in turn be interacting with each other) in urban landscapes (see Sect. 10.3.1), these kinds of synergistic interactions may be particularly common.

Other kinds of stressors may also be more common or more intense in urban areas, leading to greater susceptibility to contaminants or disease. In fragmented urban landscapes, native wildlife populations may be smaller or more isolated, leading to reduced genetic diversity (e.g., Delaney et al. 2010) and thereby potentially reduced resistance to toxicants (Barata et al. 2000). For some wildlife species, interspecific interactions may be more intense in urban areas. Dense populations of urban carnivores and domestic cats may result in increased predation on birds, small mammals, and reptiles. Additionally, interspecific competition may be more intense in habitat fragments with hard boundaries formed by development and roads (Crooks et al. 2010). Finally, increased disease exposure in urban areas (see Sect. 10.2.2.1) may increase disease-contaminant interactions. Thus far, studies examining the synergistic effects between contaminants and other stressors have been conducted mostly in aquatic systems, so investigation of the potential interactions between pollutants and other environmental stressors in terrestrial wildlife would be extremely valuable.

10.3.4 Contaminants of Concern

Although there are many compounds that could affect wildlife, here we focus on contaminants most likely to be encountered by urban wildlife. The vast majority of contaminant effects are likely unintentional, such as with industrial by-products and pharmaceuticals. With herbicides or insecticides, the use in the environment is intentional, but the effects on native vertebrates is not (e.g., the effects of the herbicides atrazine and roundup on amphibians). Some contaminants, such as rodenticides, directly target wildlife, although often the targets (e.g., old world rats and mice) are not native species. Nonetheless, the range of potentially affected species is much larger than those targeted (e.g., anticoagulant rodenticides used to control a single rodent species such as ground squirrels affect other rodents, such as woodrats (*Neotoma spp.*)).Secondary contaminant effects, such as effects on predators and scavengers, are presumably always unintentional.

10.3.4.1 Rodenticides

Rodenticides include multiple classes of poisons designed specifically to target nuisance rodent populations. Any mammalian poison can potentially be considered a rodenticide. In theory, the characteristics of effective rodenticides include a high degree of toxicity to rodents, a high degree of specificity to rodents, ready acceptance by rodents, and a high degree of reacceptance by rodents when only sublethal quantities are consumed. These characteristics are found in only a limited number of chemicals, so the length of time rodenticides have been in use is long compared with other pesticides, especially insecticides (Gratz 1973). The most popular rodenticides are anticoagulants, vitamin D poison (cholecalciferol), neurotoxins (e.g., bromethalin), and strychnine.

10.3.4.1.1 Anticoagulants

Anticoagulant rodenticides (ARs) are presently the most common method of rodent control used worldwide (Stone et al. 1999). ARs are vitamin K antagonists that deplete vitamin K1 hydroquinone, which is required for the production of certain clotting factors. Death occurs from hemorrhage, typically internal, and affected animals may exhibit weakness and altered behavior before death (Cox and Smith 1992). These rodenticides generally fall into two classes, "first-generation" (warfarin, chlorophacinone, diphacinone) and "second-generation" compounds (brodifacoum, bromadiolone, difethialone; Erickson and Urban 2004). Second-generation anticoagulant rodenticides (SGARs) were developed to overcome genetic resistance to warfarin encountered in rats in the 1970s (Petterino and Paolo 2001). Second generation ARs are highly persistent, highly toxic poisons that are nearly 100 times more potent than the parent compound (Hadler and Buckle 1992), although death from SGAR consumption can still occur up to 10 days after ingestion of a lethal dose (Cox and Smith 1992). First-generation anticoagulants, on the other hand, are less acutely toxic with shorter persistence times, and they must be consumed in multiple feedings to reach a lethal dose (Erickson and Urban 2004). Despite the ideal that rodenticides are specific to rodents, the vitamin K pathway that produces blood clotting factors is highly conserved across vertebrate taxa, including all birds and mammals, thereby rendering huge numbers of nontarget species potentially vulnerable.

Anticoagulant rodenticides pose a serious risk for both primary and secondary poisoning for native wildlife living in and near urbanized areas (Hosea 2000; Erickson and Urban 2004). While many wild animals are exposed to ARs in rural areas, levels of exposure are also high in urban landscapes. Hosea (2000) examined anticoagulant exposure in 74 animals collected from across California and found that >95 % of exposed animals came from areas with significant urban development. In lagomorphs and rodents that may be targets (e.g., ground squirrels) or nontargets (e.g., woodrats, rabbits) of applications, we found that 20 % of woodrats and ground squirrels were exposed to ARs in urban southern California (Moriarty et al. 2012). For bobcats in the same region, single-family high-density residential areas and golf courses, although they comprised a relatively small percentage of the study area, were significantly associated with exposure to ARs (Serieys 2014). Even for mountain lions, which generally avoid urban areas (Beier et al. 2010), more than 90 % of animals tested in urban landscapes in California (and >75 % of animals overall) are exposed to ARs (Poppenga et al. unpublished data). Bioaccumulation is especially a risk with SGARs because of their persistence in tissue, so nontarget wildlife contamination is becoming a widespread conservation issue (Eason et al. 2002; Erickson and Urban 2004; Gabriel et al. 2012).

Secondary anticoagulant poisoning of nontarget wildlife has been well documented in a wide range of animals around the world (Newton et al.1990; Eason and Spurr 1995; Berny et al. 1997; Fournier-Chambrillon et al. 2004; Albert et al. 2010). In the USA, exposure has been documented in urban areas in California for coyotes (Riley et al. 2003; Gehrt and Riley 2010), bobcats (Riley et al. 2007, 2010), and mountain lions (Riley et al. 2007; Beier et al. 2010) and in New York for squirrels, raccoons, deer and raptors, especially owls (Stone et al. 1999). Because small mammal species are the targets of ARs, predatory species such as mammalian carnivores and raptors are particularly at risk for secondary exposure. However, a recent study found that insectivorous hedgehogs (*Erinaceinae*) had high rates of exposure to ARs similar to those for the predators of small mammals (Dowding et al. 2010). Although the route of exposure in hedgehogs was undetermined, the authors suspected that hedgehog predation on invertebrates that had consumed anticoagulant baits was the most likely route.

If the exposure is significant enough to cause coagulopathy (i.e., the ability to clot is impaired), ARs can lead to direct mortality from uncontrolled hemorrhaging. For example, in a 9-year study of urban coyote ecology, anticoagulant toxicosis (i.e., poisoning) was the second leading cause of death, after vehicles (Gehrt and Riley 2010). Different taxa can have vastly different levels of resistance to the toxicological effects of ARs; for example, dogs are 100× more susceptible to the effects of brodifacoum (the most common SGAR residue found in our studies) than cats (Roder 2001). For species that are more tolerant of the compounds, exposure to ARs may result in sublethal chronic effects like increased susceptibility to disease, for example severe notoedric mange in bobcats (Riley et al. 2007; Serieys 2014). Causal mechanisms are being investigated, but there is evidence that exposure to ARs stimulates an inflammatory response that may explain increased susceptibility to mange (Serieys et al. 2013).

Anticoagulant rodenticide exposure has also been linked with increased susceptibility to other sources of mortality. Elmeros et al. (2011) found that for two mustelid species, those that were sampled after being trapped had significantly lower total AR residue concentrations than those sampled after vehicle collisions and predation events. Similarly, Serieys et al. (2014) found that 100% (N=5) of animals that died of vehicle collisions had detectable AR residues in their blood compared with 34% of captured animals, suggesting that recent exposure events potentially increased their vulnerability to vehicle collision.

10.3.4.1.2 Nonanticoagulant Rodenticides

Commonly used nonanticoagulant rodenticides belong to three classes defined by their mode of action: Attacking the nervous system, forming poisonous phosphine gas, or inducing hypercalcemia (abnormally high calcium in the blood). They can provide a lethal dose from a single feeding, though they have shorter toxicologically significant half-lives in the tissue of primary consumers than ARs, thus posing less risk to secondary consumers. Bromethalin is a neurotoxicant that interferes with cellular oxidative phosphorylation, causing a decrease in ATP synthesis that leads to respiratory arrest (Spaulding 1987). Death usually occurs within 2 days of ingestion. Zinc phosphide is an inorganic compound that reacts with the acid in the digestive system to generate toxic phosphine gas, inducing mortality (Rodenberg et al. 1989). Death can occur within a few hours of ingestion. Cholecalciferol is a sterol (vitamin D_3) whose ingestion induces mobilization of calcium from bone matrix into blood plasma leading to calcification of soft tissue (blood vessels, kidneys, stomach, lungs) and cardiac arrest (Jolly et al. 1993; Murphy 2002). Death can occur 3–4 days after a single feeding.

Few studies have addressed the potential hazards of nonanticoagulant rodenticides for nontarget wildlife. Zinc phosphide is thought to pose greater primary risk to certain bird species, especially those more sensitive to the poison (Marsh 1987; Brown et al. 2002). Erickson and Urban (2004) proposed that zinc phosphide was the most dangerous rodenticide for nontarget birds and mammals in terms of acute toxicity (i.e., toxicity from a single exposure). The studies performed on secondary exposure risks have been in controlled settings and indicated few adverse effects. Five studies available for zinc phosphide tested the impacts of secondary exposure on five predatory bird species, three passerines, and two foxes. Across the five studies, none of the test animals died, although Bell and Dimmick (1975) observed signs of intoxication in great-horned owls (*Bubo virginianus*) and a gray fox. For cholecalciferol, Eason et al. (2000) found a risk for hypercalcemia and kidney damage in dogs fed poisoned food, and that certain bird species were more sensitive than others, indicating potential risk in the wild.

10.3.4.2 Insecticides and Herbicides

Although insecticides and herbicides are not specifically designed to harm vertebrates, their widespread use in urban areas has the potential to cause significant impacts, especially for aquatic species. Insecticides and herbicides are widely used throughout the world in agriculture, for public health to control disease vectors (Heudorf and Angerer 2001), and in residential areas for structural pest control and home and garden uses (Hintzen et al. 2009). Organochlorines, such as DDT, were used intensively as insecticides until the 1970s when their use was prohibited in many industrialized countries. Once organochlorines were phased out, replacement compounds, including carbamates, organophosphates, and pyrethroids, were developed and are presently used extensively to control insect populations (Heudorf and Angerer 2001).

Over the last decade, several studies have demonstrated that urban use of insecticides can result in contamination of terrestrial and aquatic systems. Of particular concern is the link between the use of pesticides around residential homes and contamination of urban waterways, with impervious surfaces like concrete or pavement transmitting the contaminants through runoff (Jiang et al. 2011). Within the USA, insecticides are more frequently detected, and generally at greater concentrations, in urban streams than in streams in more rural agricultural areas (Fuhrer 1999). Organophosphates, specifically chlorpyrifos and diazinon, were once the dominant insecticides found to contaminate urban streams. However, these two organophosphates were withdrawn from most urban uses in 2001 and 2004 as a result of increased restrictions by the EPA. Pyrethroids have subsequently replaced the restricted organophosphates as the dominant insecticides found to contaminate urban streams, although little is known about the consequences of these compounds (Weston et al. 2011). Roundup, or glyphosate, is a common urban herbicide that affects amphibians, although it is the other components of Roundup, such as the surfactant, that generally affect vertebrates (rather than the glyphosate itself).

10.3.4.2.1 Organophosphates and Carbamates

Organophosphates (OP) and carbamates are widely used globally (Bondarenko and Gan 2004) in urban and suburban areas as lawn insecticides, herbicides, acaracides, nematocides, helminthicides, and fungicides (Fairbrother 1996) and have caused significant mortality in urban bird populations. Both pesticide groups are anticholinesterases, and because of the magnitude of their use, they are the chemicals most frequently associated with intoxication of domestic animals and wildlife. Acute exposure to carbamate or organophosphate pesticides is typically suggested by the discovery of animals in good physical condition with clinical signs of convulsions, pulmonary edema, and diarrhea from hyperstimulation of the parasympathetic autonomic nervous system, skeletal muscles, and the central nervous system (Fairbrother 1996; Berny 2007; Berny and Gaillet 2008). Organophosphates and carbamates include highly toxic compounds such as aldicarb, carbofuran, methiocarb, meyinophos, chlorpyrifos, diazinon, and parathion, and they are available in multiple forms including granules, pellets, powders, and liquid formulations. Compared with organochlorines and metal-based pesticides, organophosphates and carbamates degrade rapidly in the environment and so have been considered relatively safe for crops and animals (Ragnarsdottir 2000).

Significant primary exposure of granivorous birds may occur through the ingestion of coated seeds (Fairbrother 1996). Secondary poisoning is also described in predators and scavengers eating dead, dying, or incapacitated animals (Elliott et al. 1996; Wobeser et al. 2004; Novotný et al. 2011). Berny and Gaillet (2008) found that among poisoning cases, including some intentional ones, in red kites (*Milvus milvus*) across France from 1992 to 2000, carbamates were among the most commonly detected compounds. These insecticides were also responsible for a significant number of avian mortalities in US residential areas between 1980 and 2000 (Fleischli et al. 2004). A large die-off of Atlantic brant (*Branta bernicla*) on a New York golf course was attributed to a diazinon application to the grass. In central Ohio, two incidents of waterfowl exposure involving mallard ducks (*Anas platyrhynchos*) and Canada geese (*Branta canadensis*) followed the application of diazinon as a grubicide to lawns around ornamental ponds in apartment complexes (Stone and Crandall 1985). Urban streams have frequently been reported to contain OPs at concentrations that exceeded the criterion for protection of wildlife (Hoffman et al. 2000), although a federally mandated ban of diazinon and chlorpyrifos in the USA in 2001 has resulted in their reduction in some urban waterways (Banks et al. 2005a, b).

10.3.4.2.2 Pyrethroids

Pyrethroid insecticides are among the most widely used pesticides; in fact, their availability has increased since the late 1990s (US EPA 2005) to the point where they are the active ingredients in most insecticides available for residential use in the USA (Weston et al. 2005), and they pose a threat to urban wildlife, particularly aquatic species. In vertebrates, pyrethroids disrupt the normal function of the peripheral nervous system (Palmquist et al. 2012). In urban areas, these compounds are commonly used on ornamental plantings, residential lawns, and golf course turf, and for structural pest control and vector control (Kuivila et al. 2012). Pyrethroids commercially available include permethrin, cypermethrin, cyfluthrin, deltamethrin, bifenthrin, and fenvalerate (Heudorf and Angerer 2001).

In terrestrial animals, exposure to pyrethroids may occur by uptake through the skin or eyes, by inhalation, or by ingestion. Bird and mammal species are considered to be largely unaffected by pyrethroids (Palmquist et al. 2012) which are assumed to be converted to inactive metabolites by hydrolysis (Corcellas et al. 2012), but a number of recent laboratory studies have shown endocrine disruption, carcinogenicity, and neurotoxicity in pyrethroid-exposed rodents (Shafer et al. 2008; George et al. 2011; Jin et al. 2012). We know of only one published study about the impacts of these compounds on wild mammal populations. Alonso et al. (2012) found pyrethroids in the livers of all 23 male dolphins tested off the state of Sao Paulo, Brazil. Pyrethroids were also detected in breast milk and placenta samples from female dolphins, indicating that transfer to offspring could occur in utero and during nursing (Alonso et al. 2012). Given the high trophic status of cetaceans, Alonso et al. (2012) propose that bioaccumulation occurs despite the presumption that pyrethroids are deactivated by metabolic pathways. These recent lab and field studies suggest that pyrethroids may not be as benign to birds and mammals as commonly assumed.

Pyrethroids are typically introduced into aquatic systems via runoff from sprayed fields, lawns, parking lots, or during rainstorm events (Palmquist et al. 2012). They have been considered extremely toxic to fish for more than two decades (Haya 1989), with symptoms affecting the nervous, renal, and respiratory systems leading to acute toxicosis and death. Thus, the toxic threat pyrethroids pose to urban waterways has received increasing attention (Weston et al. 2005). Pyrethroids are hydrophobic organic pollutants with very low water solubility and a high affinity for soil and particulate matter (Laskowski 2002). Concrete surfaces and drainages in suburban and urban areas facilitate the transport of greater concentrations of pyrethroids into urban creeks compared to the particulate-rich runoff channeled through earthen ditches in agricultural areas (Weston and Lydy 2010). Weston et al. (2009) found that a single storm released as much bifenthrin into an urban creek in 3 h as that released over 6 months of runoff from a similarly sized agricultural watershed.

Recent studies in California, Texas, Illinois, and China have shown that pyrethroids that accumulate in urban creek sediments remain biologically available at concentrations acutely toxic to sensitive aquatic life including benthic invertebrates and fish (Holmes et al. 2008; Hintzen et al. 2009; Weston and Lydy 2010; Ding et al. 2011; Mehler et al. 2011). Kuivila et al. (2012) found that pyrethroids frequently occur in urban stream sediments in seven metropolitan areas across the USA and may be contributing to sediment toxicity across the country. Looking beyond their presence in urban waterways, much more attention should be directed to the impact these compounds may have in both aquatic and terrestrial ecosystems and on wildlife.

10.3.4.2.3 Endocrine Disruptors and Estrogenic Chemicals

Endocrine disruptor compounds (EDCs) are an important global problem highlighted first by the impacts of DDT and analogs. In 2002, the World Health Organization defined EDCs as "an exogenous substance or mixture that alters functions of the endocrine system and consequently causes adverse health effects in an intact organism, or its progeny, or (sub)populations" (World Health Organization 2002). The commonly recognized EDCs come from many different types of chemicals but include persistent organic pollutants such as DDT and analogs; industrial compounds such as dioxins, bisphenol A, and polychlorinated biphenyls (PCBs); pesticides including chlorinated insecticides, imidazoles, and triazines; chemical substances that are widely used in cosmetics such as phthalates and parabens; and several heavy metals (Roy et al. 1998; Oishi 2002; Choi et al. 2004; Carr and Norris 2006). The list of metallic EDCs includes cadmium, mercury, arsenic, lead, manganese, and zinc (Iavicoli et al. 2009). Many EDCs are found in treated sewage water and some industrial effluents (Jobling et al. 1998; Walker 2006). When absorbed into the body, EDCs either mimic or block hormones and disrupt the body's normal functions. Consequences of exposure can include reproductive impairment (Vos et al. 2000), DNA damage, and genomic instability (Roy et al. 1998). Among 48 EDCs evaluated by Choi et al. (2004), estrogenic effects were the most predominant in pesticides, while effects on thyroid hormone were found for heavy metals.

Exposure can occur as a result of direct contact with pesticides and other chemicals or through ingestion of contaminated water, food, or air. Many endocrine disruptors are persistent in the environment, and because they accumulate in fat, bioaccumulation and biomagnification within food chains can occur (Guillette et al. 1996), affecting both humans and wildlife (Colborn et al. 1993). DDE-induced egg-shell thinning was found by 1970 to have caused severe population declines in a number of avian species in Europe and North America (Ratcliffe 1970; Peakall et al. 1973; Lundholm 1997), but more recently dioxin accumulation and thyroid lesions were documented in adult jungle crows (*Corvus macrorhynchos*) in urban and suburban Tokyo (Kobayashi et al. 2005). Kime (1995) demonstrated widespread sexual disruption and developmental impacts in wild fish populations resulting from exposure to chemicals from industrial and sewage effluent in British rivers. In the San Francisco Bay Area in California, native fish had altered thyroid endocrine parameters associated with exposure to PCBs (Brar et al. 2010), and harbor seals had higher leukocyte counts and lower red blood cell counts associated with polybrominated diphenyl ethers (PBDEs), PCBs, and DDE (Neale et al. 2005).

10.3.4.3 Heavy Metals

Heavy metal pollution of ecosystems occurs as a result of human activities such as metal smelting, waste disposal, sewage treatment, use of rock phosphate fertilizers, and mining. Metals of concern to wildlife include cadmium, lead, mercury, copper, zinc, antimony, and arsenic. Exposure to heavy metals can have impacts on fitness characteristics of exposed animals (Pankakoski et al. 1993; Larison et al. 2000; Eeva et al. 2003; Gorissen et al. 2005). For example, heavy metals have reproductive effects on birds including decreased egg production, decreased hatching success, and increased hatchling mortality (Scheuhammer 1987).

Some metals are taken up by plants from the soil (Shore and Douben 1994) and transferred through the food chain more readily than others (Hunter and Johnson 1982). Metals may be taken up into the food web via a soil-vegetation-invertebrate-insectivore pathway (Hendriks et al. 1995; Reinecke et al. 2000), so insectivores such as shrews (*Soricidae*) and hedgehogs are at particular risk of exposure (Ma et al. 1991; Shore and Douben 1994; Nickelson and West 1996). Earthworms accumulate heavy metals such as cadmium and lead through ingestion of soil (Hendriks et al. 1995; Reinecke et al. 2000), leading to secondary exposure in insectivores. Pankakoski et al. (1993) found that moles (*Talpidae*) in the metropolitan area of Helsinki had higher concentrations of heavy metals, specifically cadmium, lead, and mercury, and lower body weights, than moles in rural areas.

Presently, mercury contamination is primarily a byproduct of the burning of wastes containing inorganic mercury and of coal and fossil fuels. Methylmercury, in particular, constitutes the greatest threat to fish and wildlife (Scheuhammer et al. 2007) and can be bioaccumulated and concentrated through the aquatic food web, posing the greatest risk to predatory species (Fitzgerald et al. 1998). Ackerman et al. (2008) found evidence of increased mortality in avocets (*Recurvirostra americana*) and black-necked stilts (*Himantopus mexicanus*) in San Francisco Bay. In 1989, mercury toxicosis was assigned as the cause of death of an endangered Florida panther in Everglades National Park (Roelke et al. 1991). Mercury exposure is also implicated in the decline of the rusty blackbird (*Euphagus carolinus*; Edmonds et al. 2010), one of the fastest declining land-bird species in North America (Greenberg and Droege 1999), because the bird relies on industrial wetlands and areas downwind of industrial areas for habitat.

Lead is another toxic metal known to accumulate in and have adverse effects on wildlife populations (reviewed in Eisler 1998). In the 1980s, environmental lead pollution was so pervasive it was considered to be the contaminant of number one concern to human health (Hejtmancik and Dawson 1982). Lead enters the environment through both natural and anthropogenic routes, although anthropogenic sources account for the majority of lead contamination (Johnson 1998). Wastewater-treatment plants are a significant source of environmental lead. In Illinois, it is estimated that more than 700,000 pounds of lead per year are released to the air and water from metropolitan Chicago wastewater-treatment plants (Fitzpatrick et al. 1995). Lead exposure can have serious consequences for wildlife, damaging the hematopoietic, nervous, skeletal, and excretory systems (Kendall et al. 1992;

Scheuhammer and Norris 1996). In Brisbane, Australia, Hariono et al. (1993) found that urban fruit bats had toxic levels of lead in various tissues, while lead concentrations in nonurban bats were very low. The source of the lead was not verified but it was likely atmospheric. Scheifler et al. (2006) examined whether common blackbirds in an urban area of France were exposed to lead after soil transfer of lead into the food chain. Lead concentrations in the birds and in earthworms, their principle food, were all significantly greater in urban than in rural animals. Lead concentrations in earthworms were also greater in metropolitan Helsinki compared with rural areas, and moles in urban areas accumulated lead through a diet of earthworms (Pankakoski et al. 1993).

10.4 Approaches to Managing Disease and Contaminant Impacts on Urban Wildlife

10.4.1 Disease

Once a disease is present in urban wildlife populations, little can be done to remove it, and management focuses on coping with the disease. Diseases can be prevented by developing stronger regulations limiting movement of nonnative animals between locations and requiring sanitary procedures that reduce chances of contamination. Diseases like the chytrid fungus in amphibians, for example, are thought to have been moved around by people and their pets (see Sect. 10.2). Certain procedures (e.g., bleaching or freezing clothes or equipment) can reduce the likelihood of transmission of aquatic disease between water bodies. However, in urban areas, it can be hard to create sufficient public awareness about disease issues, much less achieve compliance.

Requiring regular vaccinations for pets and keeping pets inside at all times may reduce the transmission of diseases between pets and wildlife. These actions are often recommended, but not mandated. Another option is to try to limit contact between wildlife and domestic animals, possibly with laws or ordinances requiring that dogs be leashed or prohibiting dogs in natural areas. This may be more possible in island situations, where human populations are small and travel is limited, such as on Catalina Island in California where pets are restricted to protect endangered island foxes from disease.

One potential source of disease for urban wildlife that is particularly controversial (Peterson et al. 2012) is feral cats, in particular cats in colonies or in trap-neuter-release (TNR) programs (Duarte et al. 2010; Stojanovic and Foley 2011). The TNR programs generally involve vaccinating and neutering colony cats, and proponents of the programs argue that this can decrease the chances of disease transmission to wildlife. In practice these programs can be expensive (Lohr et al. 2013), it can be hard to capture all of the cats, and vaccinations do not last forever, so animals would need to be recaptured to maintain the protection. In many cases, new cats are dropped off at these colonies, which can increase the possibility of susceptible cats. Anyplace where animals are regularly being fed outdoors, either unintentionally or intentionally and whether it is pets, feral cats, or wildlife such as raccoons or coyotes, there is an increased potential for disease transmission. Hence, cat colonies or TNR programs in or near natural areas should be particularly scrutinized for disease concerns.

It is likely that urbanization will continue to be implicated in the emergence of current or future emerging infectious diseases (EIDs), with potentially significant consequences for wildlife populations, natural ecosystems, and human health (through exposure to zoonotic diseases). Future research should focus on disease exposure in urban areas, including the ways in which anthropogenic activities influence the modes of disease transmission and levels of host susceptibility.

10.4.2 Contaminants

Suburban and low-density urban areas, with their relatively rich wildlife faunas coupled with extensive and varied contaminant use, may be especially ripe for contaminant exposure and effects, as well as research on contaminants and wildlife. The public is largely unaware of the extent to which wildlife populations are exposed to and can potentially be affected by contaminants. Outreach through the media, public presentations, websites, and social networking can help people understand these impacts (Chap. 11).

One important consideration for contaminants is off-label use, particularly for those that are intentionally put into the environment (e.g., insecticides, herbicides, and rodenticides) but that may be affecting nontarget species. For example, second generation anticoagulant rodenticides are intended only for use indoors or immediately adjacent to buildings, targeting rats and mice inside buildings, and only first generation ARs, which are far less toxic and persistent, are to be used outside. However, we, and many others, find extensive exposure to these SGAR compounds in wildlife, suggesting extensive off-label use (see also Bartos et al. 2012).

The most direct way to affect contaminant use is through policy action, which can occur at the national (the Environmental Protection Agency), state (state Departments of Pesticide Regulation), county (through county councils or boards of supervisors), or city (city councils) levels. In some cases, the unintended effects of contaminants on nontarget wildlife, such as the herbicide atrazine causing the demasculinization of frogs (e.g., Hayes et al. 2010) and the lethal effects of the extremely common herbicide Roundup on amphibians (Relyea 2005) have generated interest and concern in both agencies and the public.

Information from wildlife studies, including our work in southern California, on the impacts of contaminants on nontarget carnivores was used by the EPA in 2008 to create new regulations that limit second-generation ARs such as brodifacoum and bromadiolone (see Sect. 10.3.4.1). The regulation limits their application to bait boxes, but much more importantly limits their sale to only licensed providers. In early 2014, the state of California enacted similar regulations. The effect of these regulations may be limited by the use by licensed providers, specifically pest-control operators, that may be a significant portion of total use (e.g., Steinberg et al. 2014), so widespread exposure may continue.

There have been local changes in policy relative to the use of these pesticides as well. After the deaths of two mountain lions, a local county supervisor in Ventura County instituted significant changes and reductions in the use of ARs by county personnel and on county property. And, citizens and activists in the region have taken local action, including lobbying city councils (e.g., Malibu, Calabasas) to pass resolutions recommending against the use of second generation anticoagulants and working directly with various businesses in the area to stop selling ARs. Similar efforts have occurred in cities in other parts of the state, largely in the San Francisco Bay Area, and Marin County. One factor in the public's mobilization around this issue is that it is an occasion where direct action can be taken by individuals. In other words, it is difficult to have much effect on urbanization or habitat fragmentation, but individual action and activism may make a difference in the methods and degree of contaminant use.

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Chapter 11
Urban Wildlife Communication and Negotiation

Susan K. Jacobson, Dara M. Wald, Nia Haynes and Ryo Sakurai

Keywords Attitude · Audience · Behavior · Communication system · Citizen science · Collaboration · Conflict resolution · Mass media · Negotiation · Outreach · Social media · Source · Stakeholder · Target audience

11.1 Introduction

Communication shapes how urban audiences affect wildlife management in many ways, ranging from policy making and citizen science to conflict management. In the USA, citizens can vote directly on policy making initiatives (Initiative and Referendum Institute 2012), which has broad implications in the field of wildlife management. For example, Florida residents voted to support "Florida Forever," the largest conservation land-buying program in the nation. On the other hand, Michigan voters rejected a ballot initiative that would have allowed hunting of mourning doves (*Zenaida macroura*), despite support for the program by the Michigan Department of Natural Resources (Wildlife Management Institute 2012). Urban audiences offer a large pool of volunteers for citizen science projects, including efforts to monitor species in urban areas. From 2001 to 2010, volunteers in Chicago and New York City were able to observe and record more than 86% of the estimated number of butterfly species present in these cities (Matteson et al. 2012). Data collected and shared electronically by bird-watchers through the BirdSource program (www.birdsource.org), a collaborative website of the National Audubon Society and the Cornell Lab of Ornithology, helped New York Audubon chapters to address conservation issues at the Jamaica Bay Important Bird Area (IBA) in Queens, and to identify the Montezuma Wetlands complex as an IBA, enabling conservationists to obtain federal funding for habitat acquisition and restoration (Fitzpatrick and Gill 2002). None of these initiatives would have occurred without communication with a diverse and engaged public.

S. K. Jacobson (✉) · D. M. Wald · N. Haynes · R. Sakurai
Department Wildlife Ecology and Conservation, University of Florida, Gainesville, FL 32611-0430, USA
e-mail: jacobson@ufl.edu

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Communication can help reduce conflicts over wildlife management. In some cases, communicating with the public can increase human tolerance for wildlife, and be more cost-effective than spending money on relocating "nuisance" species, such as bears near urban areas (Anderson 2007; Peterson et al. 2010). Communication also can be used to build support for wildlife management and to educate people about wildlife. In New Hampshire, a "Learn to Live with Bears" communication campaign used messages about bears and food to change human behavior in ways that effectively reduced nuisance bear behaviors (Organ and Ellingwood 2000). Project Coyote (*Canis latrans*), a wildlife advocacy organization in Los Angeles County, California, launched a campaign to educate urban residents on "coyote hazing" techniques (e.g., banging pots and pans, opening umbrellas, screaming) in an effort to reduce negative human–coyote interactions (Mendoza 2012). The US National Park Service has an entire division dedicated to "consultation, cooperation, and communication" with the public, and regularly holds public and stakeholder meetings to build support for management initiatives (National Park Service 2012).

Despite these efforts, there are many examples of failed management initiatives caused by poor communication or lack of communication that resulted in conflict among stakeholder groups or between stakeholders and managers. Stakeholders (i.e., groups that have an interest, or stake in a wildlife issue or management concern) can make or break a new wildlife management initiative and influence public policies toward wildlife and natural areas. Program success can be directly related to stakeholder satisfaction with a program and/or support for program goals (Ford-Thompson et al. 2012). Communication is especially important in urban areas where stakeholders are numerous and human–wildlife interactions are frequent. Management programs conducted on public land must undergo some form of public review (Warner and Kinslow 2011). Habitat conservation can be more complex in urban areas where an array of city, county, state, and federal agencies may have jurisdiction over management of wildlife and ecosystems. Moreover, laws and regulations, such as restrictions on the use of leg-hold traps or firearms within the urban area can constrain management options. An initiative by the Maryland Department of Natural Resources to control the local population of invasive mute swans (*Cygnus olor*) was hampered by vocal public protests and a lawsuit filed by the New York-based Fund for Animals (O'Connell 2003). Effective communication is critical to reducing and preventing such conflict. To communicate effectively, wildlife professionals must consider the content of the messages they are sending, how they are sending them, who is listening to the information being provided, and how they are receiving it.

In addition to addressing traditional constituents, such as hunters and anglers, wildlife agencies must embrace new and increasingly vocal nonconsumptive stakeholders, from birdwatchers to animal welfare groups. Urban constituents are becoming more technologically savvy, more diverse, speaking different languages, and viewing wildlife through different cultural lenses. These changes require wildlife professionals to use new skills and techniques for effective outreach and communication.

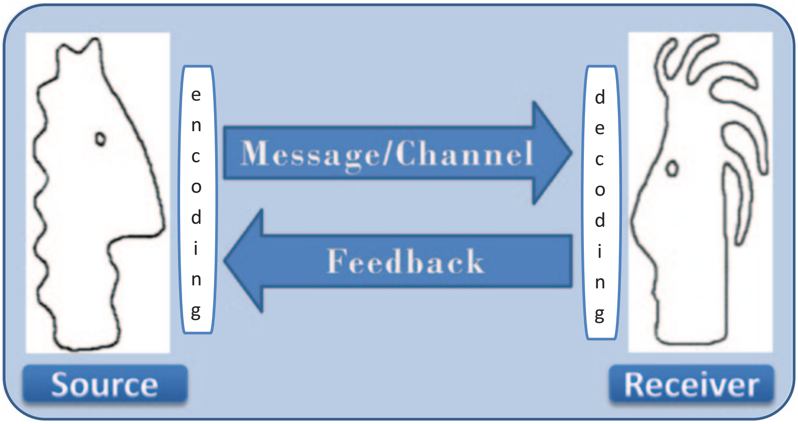

Fig. 11.1 A basic model of communication reveals the complexity involved in delivering a message about wildlife issues (Jacobson 2009, p. 11)

Urban wildlife management requires the use of both mass media and interpersonal communication to solve pressing problems. Given the potential impact that effective communication programs can have, it is necessary for urban wildlife managers to understand the basics of communication to increase their efficacy as professionals. The goals of this chapter are to: (1) describe the communication process for identifying communication objectives, targeting audiences, selecting channels and messages using mass media and social dialogues, and evaluating results; and (2) present strategies for wildlife communications to help negotiate conflicts among diverse urban audiences.

11.2 Elements of the Communication Process

This section identifies the elements of communication and provides examples of communication tools that can help alleviate common communication breakdowns. In the second section, communication involving stakeholder groups and negotiation are discussed. Communication is a process of exchanging ideas and imparting information. Communication involves conveying a message that is understood as well as actively listening to others. A message, verbal, visual, or written, that is not understood by the receiver is not communication. The only way to ensure the message is received and understood is to incorporate feedback into the communication process. Understanding a variety of ways to communicate can help ensure that wildlife management goals and objectives are achieved and conflict is minimized. Knowledge of the components of the communication system—the source, message and medium, receiver, and evaluation loop—can help wildlife professionals design effective communications. Feedback from the audience allows professionals to assess and improve their objectives and strategies (Fig. 11.1).

11.2.1 Source

The communication source may be a person, organization, or agency spokesperson. For example, a scientist explains the importance of the discovery of a new leopard frog species (*Rana pipiens*) in New York to media representatives (Foderaro 2012). For environmental issues, the credibility of the source usually involves public perceptions of the source's expertise, trustworthiness, and power. Sources may know how they want a message to be received, but cannot control how the message is spread by social media, encoded by media gatekeepers, or decoded and interpreted by the receiver. Agency sources are dependent on gatekeepers who regulate the flow of information to receivers and may introduce or remove information contained in the communication. For example, a news editor decides whether to accept or change a press release submitted by a wildlife agency. Choosing a source from among stakeholders groups can help ensure that the cultural context of the message is on target and that an empathetic perspective is presented. Alternatively, well-known figures can be effective sources. Using celebrities as sources can help disseminate a message directly to broad audiences outside typical gatekeepers that interact with agencies. People for the Ethical Treatment of Animals (PETA) has captured media attention with their advertisements featuring celebrities opposing the use of fur and animal skin. These feature celebrities touting slogans such as "Ink, not mink," "Whose skin are you in?" and "Give fur the cold shoulder" (People for the Ethical Treatment of Animals 2012). Bette Midler, a singer and actress, founded the New York Restoration Project (NYRP), an initiative to restore and maintain green spaces in New York City (New York Restoration Project 2012). However, while celebrities can spread messages widely, researchers have found that messages promoted by celebrities who have little experience with an issue may engender skepticism (Cammisa 2009). Perceptions of source credibility and how the message is delivered will affect how the receiver decodes the information. Both encoding and decoding are important parts of the communication process for all types of media, from a speech or brochure to a web-based interactive site.

11.2.2 Message and Medium

The goals and objectives of wildlife agencies or organizations should guide selection of the message and medium. For example, groups may have an interest in the goal of a communication program to control nonnative invasive species, conserve land, restore a city park, or protect threatened wetlands. The situation and message will dictate the optimal media approach for achieving specific objectives. Communication objectives may target changes in the audience's knowledge, attitudes, or behaviors. An initial message may try only to increase awareness about a wildlife issue. Further objectives may focus on shifting attitudes, increasing concern, and finally promoting action. The overall goal may be to accomplish tasks associated with organizational missions, such as gaining protection for a particular wildlife species or raising public awareness of an issue to influence legislative actions. Whatever the medium—print or electronic—it is important to keep communication objectives in mind. To be effective, the message must be interesting, informative, and persuasive.

For example, your objective might be having 100 residents attend your wetland Bioblitz and rapidly identifying amphibian species inhabiting an urban pond. A more ambitious objective would be to change the behavior of participants, for example, convince 10% of the participants to provide funding or lobbying for wetland restoration initiatives. Alternatively, the objective may be to recruit volunteers for a long-term monitoring program to aid state biologists in evaluating changes in amphibian populations. This objective would attract participants with different motivations and would aim to develop different skill sets among the participants (Jacobson et al. 2012).

The message must be crafted with the objective in mind. Many outreach programs are designed simply to increase awareness about an organization, resource, or site. In New York City, The Nature Conservancy (TNC) collaborated with the New York Foundation for the Arts to increase awareness of urban ecology through a hands-on art experience (Jacobson et al. 2006). Their outdoor program, Wild New York, created a field guide for urban environments, provided a naturalist and an artist guide to accompany participants, explain ecological phenomena, and introduce a new art project to four natural sites. The program was advertised by the Foundation for the Arts, reaching new audiences for TNC that had never attended their traditional ecology-based programs, and who reported new interest in visiting and appreciating natural areas.

Once the objectives of a communication campaign are determined, the ideas are encoded and transmitted in the form of a message. Often wildlife management stories do not get transmitted by news media because they are viewed as too complex, long term, or unimportant (Corbett 2006). Simple messages and current events are understood most easily by journalists and media gatekeepers as well as the public. One example of a simple and direct message to prevent the public from feeding bears is “A fed bear is a dead bear,” promoted by the New Hampshire Fish and Game Department. Messages communicating complex issues are harder to transmit to the public. The issue of prescribed burning as a management tool in the wildland–urban interface, for example, has been challenging to translate into a simple message. In a survey of West Florida residents, only 12% knew that regular fires are a natural process maintaining their native pine forests (Jacobson and Marynowski 1997). The Division of Forestry’s slogan, “Rx Fire: Prescription for Forest Health,” seems difficult to say, much less understand. Coverage of wildfires and benefits of prescribed fire in the media can stimulate public interest in prescribed burning as a valuable conservation tool (Jacobson et al. 2001).

Addressing the needs of an audience can help in developing appropriate messages to influence behaviors. All people are motivated by various needs and desires. The psychologist Abraham Maslow (1954) developed a simple hierarchy of needs. He suggested that people first fulfill their physiological needs for food, health, safety, and security, before addressing their need for a sense of belonging to a group, self-esteem, and ultimately, self-fulfillment. The value of clean water is central to basic needs for human health and providing habitat for wildlife. Nonmaterial uses of natural resources address people’s need for a sense of belonging and include the value of nature for religious beliefs, educational uses, physical and emotional fitness, and recreation. Social motivations were found to be important to volunteers for the Florida Fish and Wildlife Conservation Commission. The opportunity to be part of a group was almost as important as the opportunity to help wildlife, and needed to be mentioned in program recruitment messages (Jacobson et al. 2012).

Many disciplines, including psychology, education, and sociology, contribute to communication theory. Increasing public knowledge alone does not guarantee conservation action. Other factors, such as the perception of one's ability to affect change and commitment toward conservation, also influence behavior (Kollmuss and Agyeman 2002). Understanding where the audience is in the adoption of a new behavior can help better target a message. For someone who is still thinking about an issue, the message should promote positive outcomes and expectations. Someone who has already decided to change their behavior would be better served with a message that details how to correctly conduct the behavior (Jacobson et al. 2006). Social influences or norms can be a powerful force driving the adoption of environmentally responsible behavior (Fishbein and Ajzen 1975; McKenzie-Mohr and Smith 1999). Support of a new technology by a family member or the introduction of a new concept by a friend influences others to consider new behaviors. The influence of social norms has been harnessed in campaigns to reduce littering in city parks to maintaining wildlife friendly yards. The National Wildlife Federation uses attractive yard signs to appeal to people's interest in social recognition and spread their message to promote wildlife friendly landscapes (National Wildlife Federation 2013). Choosing a medium is as important as choosing message content, and different channels present different advantages. Carefully pilot testing messages and media on the intended audience is a critical step in developing an effective campaign. Communication involves both interpersonal and mass media approaches. Interpersonal communication involves conversations, group meetings, speeches, and participatory demonstrations. Mass media includes the internet, newspapers, television, mail, films, and publications.

Selecting the appropriate communication medium based on goals and the target audience is critical. Different types of media transmit messages of varying complexity to different audiences (Table 11.1). Mass media may be effective in setting the public agenda, building awareness, and reinforcing opinions (Jacobson 2009). The public receives much of its environmental information through mass-media channels, with the use of the internet increasing to over 71 % of American homes (U.S. Department of Commerce 2011). Interpersonal and hands-on activities can be more effective in influencing attitudes and behaviors than mass media (Jacobson et al. 2006). Van Heezik et al. (2012) found that person-to-person dialog was the best method of encouraging urban residents to plant wildlife friendly gardens. The traditional top–down approach of communicating scientific information did not work as well as a dialog that gave the homeowners a sense of free choice and self-motivated learning to facilitate behavior change.

To reach a larger audience and strengthen the influence of a message, it also is effective to use more than one media channel. The American Bird Conservancy has launched a campaign to encourage pet owners to keep their cats indoors, citing risks to wildlife from cat predation and injury. The campaign includes a website,

Table 11.1 Functions and constraints of various media for spreading a wildlife message. (Adapted from Jacobson 2009, pp. 251–252)

Purposes	Constraints
Press interview	
Identify your organization as an authority	Advance preparation needed
Allow a personal appeal for a campaign or project	Difficult to arrange
Provide human interest	Requires good speaking skills
Provide wide exposure	
Advertisement	
Deliver a specific message	Expensive
Control the time, duration, and place message will be seen or heard	Need for technical skill
	Delivers only brief, simple messages
Public service announcement	
Advertise free activities, events, or materials	Little control over time, place and duration
Highlight a wildlife issue	Competition for space and airtime is fierce
Recruit volunteers	Need technical expertise to meet standards of press or broadcast station
Inexpensive exposure	Delivers only brief, simple messages
News release	
Announce hard news, such as an action, award, new study, or new leader	Competes for attention with other organizations
Alert the public to an event or benefit activity	News must have broad interest and be timely
Reach many media outlets and the public at once	Media contacts may not understand newsworthiness of issue
Press kit and fact sheets	
Provide background information on a wildlife project or issue	Must be up-to-date
Provide supporting material for reporters at a news conference	Seldom useful alone
Augment other media materials by providing specific descriptions of projects or activities	Audience must have prior interest to use fact sheets
Summarize data regarding wildlife programs or events	May provide excess information
Press conference	
Break a news story quickly	Advance preparation essential
Provide information to many outlets if interest in story is already generated	Complex logistics
Garner interest if a participant in the event has personal news value, such as a celebrity or local expert	Chance of low interest and turnout
Announce significant news (e.g., dedication of new land, new wildlife policy, corporate donation, or wildlife initiative)	Requires public speaking skills
Letter to the editor	
Clarify your position	Information can be viewed as biased or frivolous
Voice a disagreement with a newspaper editorial or community policy	

Table 11.1 (continued)

Purposes	Constraints
Thank community members bring attention to a specific cause	
Website and electronic mail	
Inexpensively reach a potentially large audience	Accuracy of information presented may be questioned
Provide and update information quickly	Need expertise to set up effective Website
Make a lot of information accessible to interested viewers	Requires funding to purchase domain or build site
Distribute materials 24 h a day	E-mail requires a reliable list of stakeholders
Offer a wide variety of electronic media over one medium	
Social media—facebook and twitter	
Email, chat, and video chat functions integrated	Privacy concerns
Spread information to like-minded audience	Difficult to control message and content
Information source and part of conversation for latest news	Requires continuous attention
Network and connect with an international user base, particularly younger audiences	

downloadable fact sheets, printable brochures, DVDs, 30-s radio public service announcements (PSAs), posters, sample letters to the editor, teachers' aids for K-6 classroom education, and videos about cat colonies (www.abcbirds.org). Before developing a communication campaign, it is important to assess which medium will reach the audience and help achieve the desired outcomes.

11.2.2.1 Using Mass Media and Technology

Mass communication and social media can be used to communicate urban wildlife management issues (e.g., Jacobson 2009). In 2008, two red-tailed hawks took up residence on a window ledge of the Franklin Institute's building in Philadelphia, PA. Recognizing public interest in urban wildlife, the institute placed a camera on the ledge, which streamed video of the hawk nest and established the Hawkwatch blog to follow the exploits of these urban hawks and their young (The Franklin Institute 2012). The institute learned from the public outcry generated when a luxury co-op building in New York City destroyed the nest of Pale Male and his partner (an unusual breeding pair of red-tails in Manhattan, made famous in a book and film). This negative publicity from New York led to the petition by the Wild Equity Institute (2012) to the US Fish and Wildlife Service to change its migratory bird nest policy which allowed the destruction of unoccupied nests without a permit.

New technologies can help bring wildlife to urban audiences. Both youth and adults, who influence the use of public and private lands, communicate on web-based media such as YouTube and Facebook. Youth ages 8–18 spend an average of 7.5 h a day engaged in some form of media (Rideout et al. 2010). Wildlife organizations must get their messages online in creative ways to keep communications relevant, accessible, and timely. Virtual outreach has the potential to reach large numbers of people and enhance stakeholder involvement in management decisions. Hence, nongovernmental organizations, including The Nature Conservancy and the World Wildlife Fund, have enhanced their online presence by developing Twitter, Facebook, and other social media sites. Government agencies also have recognized the need to use social media; the US Fish and Wildlife Service has an entire web page dedicated to social media, including a YouTube channel, Flickr stream, and Facebook page. With a click of a button, users can access information by region (Midwest, Northeast, etc.) or by special topics, such as climate change (U.S. Fish and Wildlife Service 2012). Social media and web-based communication have extended the notion of a single source and allowed "regular" people to become spokespeople for a variety of issues. Groupon, Grassroots, and change.org enable individuals to start petitions online about issues they consider important. Through these sites, supporters can link to friends via Facebook or Twitter who might be interested in urban wildlife, download petitions, donate money to an organization, find important contact information of politicians or managers, and generally mobilize support. The Crowdrise website provides urban audiences with a selection of fundraising opportunities from urban garden programs to African elephant protection. In addition, blogs have become a popular tool for communicating urban environmental issues; examples include "The City Birder" focused on urban birds and wildlife around New York City (citybirder.blogspot.com) and the "Urban Wildlife Guide" that provides information and stories about insects and wildlife (urbanwildlifeguide.net).

In addition to developing web-based campaigns, new technological developments allow citizen science projects to harness data collected by stakeholders and citizen groups to help advance scientific knowledge (Bonney et al. 2009). Citizen science programs serve to inform the public about the specific organisms or processes they are observing while helping scientists develop and implement projects that yield both scientific and educational outcomes. Developing effective communication to attract or involve urban residents in wildlife science requires standardized protocols for providing or collecting data, training and recruitment plans, and collaborating on overall goals to address issues that are of concern or interest to citizens as well as scientists. The Fort Worth Zoo in Texas has developed the "Scales on Trails" field project to estimate the urban snake population in the zoological park. Volunteers can help zoo employees identify snakes and collect basic information about the location of the snake and its health. In addition, each animal is tagged so that information about the snake's movement, growth, and lifespan can be recorded (Fort Worth Zoo blog 2012). In Yokohama, a Japanese city with more than 3.5 million residents, 23 years of observation by citizens revealed the effects of climate change on the phenology of winter birds, demonstrating the value of an urban citizen science project (Kobori et al. 2012). The Cornell Laboratory of Ornithology currently runs several large citizen science projects that rely on web-based communications. Participants in the Great Backyard Bird Count gathered more than 100,000 checklists that included 623 bird species and provided critical information for scientists about avian migration patterns and climate variation.

A number of phone applications have been developed for mobile devices that encourage participation in collecting and sharing wildlife information. eBird, a well-known online repository for bird observations, has recorded more than 3.1 million bird sightings across North America. It launched a cell phone application including images and audio for more than 470 commonly seen species that allow users to upload new observations in real-time (ebird.org, Bell, personal communication 2012). SciSpy was created by the Science Channel to encourage citizens to upload pictures and information of natural phenomena that is added to existing databases to provide valuable information about migratory routes, changes in the natural environment, and seasonal variation (Sully 2011). NatureFind guides users to parks, zoos, gardens, and nature centers to encourage urban residents to participate directly in environmental activities and monitoring. These applications promote greater awareness among audiences, particularly those who already have an interest in urban wildlife and their habitats.

11.2.3 Receiver

An understanding of the audience (the receiver) is critical to designing messages and selecting media to produce effective communication. It helps ensure that the message is decoded as intended. Audience research provides one way to evaluate how receivers decode messages and the degree to which the intended message is received by an audience. Audience research involves using public surveys, focus groups, observation, internet databases, case studies, and networks with organizations that serve the audience. To help agencies tailor messages to audience needs, wildlife professionals must make use of demographic data, sociological profiles, and knowledge of public attitudes and current behaviors as described below.

11.2.3.1 Diverse Audiences

Across urban areas, communicating with stakeholders about wildlife management may involve multiple languages and cultures. The USA population is growing more ethnically and racially diverse, and nearly a quarter of the citizens identify themselves as African American, Hispanic, Asian and Pacific Islander, or American Indian (U.S. Bureau of the Census 2000). By 2050, this number is expected to exceed 47% of the US citizens with more in a number of states and urban areas (U.S. Census Bureau 2011). Minorities are often underrepresented in visitor populations at nature parks and other wildlife venues (Floyd 1999; Taylor et al. 2011). However, as demographics change, these groups represent important stakeholders in decisions about wildlife management in urban, suburban, and rural areas. Understanding differences in stakeholder cultures (i.e., values, attitudes, and behaviors) is essential in implementing public outreach programs (Lopez et al. 2005).

Researchers have demonstrated cultural differences in outdoor use, wildlife values, and environmental behaviors. When compared to Whites, Hispanic Americans visit parks in larger groups with extended families and emphasize the social benefits of outdoor recreation, such as being with family or sharing experiences with friends (Floyd 1999). There are also sociocultural connections between attitudes toward and values of wildlife. Peterson et al. (2011) suggested that differences between attitudes about wildlife ownership on private lands between Latino Americans and Whites may stem from sociocultural history of land ownership and entitlements. With regard to participation in conservation and environmental activities, studies have demonstrated that African Americans show greater affiliation with volunteer organizations that concentrate on more grassroots issues than with traditional environmental groups (Mohai 2003; Taylor 2000). Other studies have suggested that minorities offer more support for issues that have direct bearing on quality of life and that some groups are reluctant to participate in environmental activities they believe have little chance of effecting meaningful change (Johnson et al. 2004; Whittaker et al. 2005). Communicators can target the needs and desires of diverse audiences to decrease barriers to engage in wildlife related activities. For some groups, it may be more effective to focus on motivations and attitudes tied to racial or ethnic identities than directly appealing to environmental stewardship or conservation motivations (Marcus et al. 2011).

Developing communication programs is an iterative process, and objectives and activities evolve with greater understanding and involvement of audiences. Audience demographic changes affect the selection of communication media and messages. Knowledge of the target audience is critical to promote a dialogue with non-traditional groups. Media use varies among stakeholder groups (Loker et al. 1999), and residents in different locations consume different environmental information (Sakurai et al. 2012). Research suggests that ethnic media may be one form of reaching and communicating with diverse publics (Crano et al. 2008), as ethnically based sources of information were judged to be more credible than sources not tailored to the ethnic groups (Winter et al. 2008). Further, Winter et al. (2008) found that African Americans and Latin Americans relied on relatives and community groups, such as churches, as significant and trusted sources of information. These findings can be used to tailor media outreach to increase minority participation in wildlife programs. Historically, the environmental movement has been criticized for being a movement largely of middle class or more affluent Whites that ignored issues in the urban environment where many people lived (Stevenson 2007). As urban demographics change, communication about the environment must be of interest and accessible to a diverse audience (Hudson 2001). For example, the Outdoor Afro is an online blog and networking site devoted to connecting minorities with nature and the outdoors (outdoorafro.com). The website uses social media to connect members to volunteer opportunities, recreational programs, and other members who share the same interests. With over 10,000 views per month and at least 1700 Facebook friends in 2010, the site has successfully incorporated social media technology and targeted communication to traditionally underrepresented groups to create interest in wildlife and outdoor recreation related issues (MacDonald 2010). Additionally, the Earth Tomorrow program of the National Wildlife Federation works directly with urban minority youth to engage them in conservation action and service learning projects aimed at improving urban communities and wildlife habitats. This program specifically targets urban youth in an effort to enhance their involvement in environmental issues (Rogers 1998). This program illustrates how communication can be used to overcome barriers and involve diverse stakeholders in urban wildlife management. By working directly with urban youth, the program effectively reaches diverse stakeholders and encourages them to participate in conservation.

11.2.4 Feedback

Using mass communication successfully requires an understanding of what makes activities or events newsworthy. Activities that are socially useful, fill a public need, or are just plain interesting make the news. To be newsworthy, a story must appeal to people's concerns and desires. Monitoring feedback can help communicators evaluate whether they have reached an audience and addressed audience interests. This helps avoid common problems such as targeting the wrong audience or using an irrelevant message. Systematically collecting feedback allows you to assess the outcomes of your communications activity—press coverage, audience members contacted or involved, and impacts on natural resources—to identify what did and did not work.

There are various methods for collecting feedback and evaluating communication programs, including formal before-and-after surveys of audience members, and direct observations of the audience or their impacts on wildlife populations. For example, to evaluate a communication campaign to restore a wetland, you might measure increases in public awareness, count new members, or monitor changes in legislators' votes or changes in wetland protection. Near Washington DC, a pre- and post-program experimental design revealed that a communication program positively influenced beliefs and attitudes toward coyotes (Draheim et al. 2011). In Alaska, monitoring of a communications program to increase public awareness about bears was credited with the acceptance of a strict ordinance regulating refuse storage and collection (McCarthy and Seavoy 1994). The Monterey Bay Aquarium monitors the outreach activities of their Seafood Watch program promoting sustainable fisheries by helping citizens make informed choices about the seafood they eat. Their pocket-size guide and detailed website helps consumers support fisheries that are healthier for ocean wildlife. Analysis of website use can reveal popular or ignored content areas and navigation patterns to assist with improving the site (Jacobson 2009). Feedback is essential to evaluate whether a program's objectives are achieved, and whether they are able to increase knowledge, shift attitudes, or change behaviors. Feedback is the primary way to identify improvements needed in a communication campaign.

11.3 Working with Stakeholder Groups to Reduce Conflicts

Wildlife agencies and conservation organizations should collaborate with a diverse set of stakeholders who are affected negatively or positively by wildlife management (Schusler et al. 2000; Carpenter et al. 2000).Wildlife managers are tasked with balancing the competing interests of different groups and wildlife populations in light of stakeholder desires and tolerance and the biological limits of the environment (Riley and Decker 2000; Chase et al. 2000; Kellert et al. 1996). Engaging stakeholders, particularly urban communities, involves aligning agency goals with community priorities and incorporating multiple kinds of knowledge to encourage acceptance and support of management objectives (Pandya 2012). Bat Conservation International (www.batcon.org) effectively harnessed urban tourism interests and overcame negative attitudes about bats in their outreach program to protect bats in Austin, Texas. Other groups such as Central Park Conservancy (www.centralparknyc.org) focus on specific urban or regional areas.

Failure to involve stakeholders in the decision-making process can result in delays or project failures. For example, the Florida Fish and Wildlife Conservation Commission's attempt to eliminate exotic rhesus monkeys (*Macaca mullatta*) from Silver Springs of Ocala was blocked by an activist group, "Friends of the Silver River Monkey" (Floethe 2010). Efforts to eradicate the invasive Formosan macaque (*Macaca cyclopis*) in Japan were stymied in the face of strong opposition and threats from animal right organizations (Murakami and Oi 2007). Researchers have shown that stakeholders in urban environments are more likely to hold moralistic and humanistic values toward wildlife than rural residents, and therefore are more likely to oppose lethal management (Kellert 1996). This can contribute to conflict between stakeholders and managers over efforts to control urban wildlife using lethal techniques. This can be exacerbated when groups brand or rename exotic animals with local names, such as "Silver River Monkeys," effectively employing good communication skills by using language to tie exotics to a local place and imply shared ownership or responsibility.

Strong opposition by hunters to the feral pig control efforts by the Hawaii State Division of Forestry and Wildlife led to the development of the Natural Areas Working Group made up of environmentalists, community activists, hunters, and representatives of local conservation and game departments. The Working Group developed a list of management recommendations that were eventually adopted and implemented by the State of Hawaii House of Representatives with widespread public support (Josayma 1996). Involvement of target audience members and other stakeholders in the planning process helps ensure relevancy, and promotes a commitment to the implementation and long-term sustainability of the program. A diversity of partners can provide new perspectives, creative ideas and solutions, and broader bases of support, which enhance program development. Working in groups helps to share the workload and can generate additional resources to assist with labor or funding. For example, involvement of landowners in planning a workshop that educates landowners about wetland resources, identifying critical habitat, or removing invasive species helps ensure that content and format will be appropriate for other landowners who participate in the program (Demers et al. 2012).

Despite the benefits of working with groups, such collaborations are not easy. Partners may have different backgrounds, experiences, and interests. This is especially true in reaching out to diverse urban audiences through unconventional channels, such as communities of faith, art establishments, and garden clubs. Stakeholder groups have different levels of expertise and resources; conflict can be caused by strongly held and entrenched values, attitudes, or perceptions. Conflicts over wildlife and natural resources are complex and existing regulations or policies may inhibit flexibility and options. Even in situations where stakeholders experience similar issues, live with similar wildlife, and inhabit the same area, tolerance can vary both within and among stakeholder groups (Decker and Purdy 1988; Schusler et al. 2000).

Conflict between stakeholder groups and wildlife management agencies over the appropriate management method can result in legal action against wildlife management agencies, citizen ballot initiatives, delayed management action, and increased tension and distrust between managers and stakeholders (Schusler et al. 2000; Chase et al. 2000; Perry and Perry 2008; Manfredo 2008). If effective negotiations do not take place, conflicts worsen, communication stops, and crises and unwise management decisions often ensue. Wildlife professionals that understand the steps of negotiation and can communicate effectively with stakeholders are likely to more rapidly and successfully achieve wildlife management goals and objectives and avoid conflict among stakeholders. Formal mediation processes are more expensive, but may be necessary to manage intractable conflicts.

11.3.1 Conflict Resolution Activities

Stakeholders can work toward conflict resolution using several tools, including negotiation, mediation, arbitration, and litigation. Litigation can be expensive and takes control of the issues, process, and outcome away from stakeholders, so they often are willing to try alternatives. Negotiation provides stakeholders the most control over issues, process, and outcomes, though facilitators are often needed to prevent breakdowns in the negotiation process. Negotiation is a two-way communication designed to reach an agreement between two people or groups with shared, opposing, or different interests (Fisher and Ury 1991). Negotiation can be used to foster dialogue among stakeholders, improve the information on which a decision is based, build a broad base of support for an issue, and resolve controversial issues in ways that all parties find acceptable. Many books provide training in negotiation practices. In *Getting to Yes: Negotiating Agreement Without Giving In,* Fisher and Ury (1991) provide a four-step blueprint for principled negotiation. We demonstrate the four steps used to facilitate negotiation between opposing groups, drawing on examples from a common environmental and animal welfare issue in many urban areas, the management and treatment of free-roaming domestic cats. Estimates of the number of free-roaming and feral cats vary from 25 to 100 million in the USA (Centonze and Levy 2002; Dauphine and Cooper 2008). Feral cats pose a potential risk to wildlife through predation and injury, risks that have been broadly addressed by conservation biologists (Baker et al. 2008; Beckerman et al. 2007; Coleman et al. 1997) and veterinarians (Barrows 2004; Jessup 2004; Levy and Crawford 2004). The current risk of zoonotic disease spread to humans and wildlife from feral cats is not well understood; however, cats are often cited as a source of rabies, feline leukemia (FeLV), feline panleukopenia (FPV) disease, *Toxoplasma gondii*, and *Giardia* spp, infectious diseases potentially harmful to both people and wildlife (Dubey et al. 1970; Nutter et al. 2004). There are significant differences between stakeholders in perceptions of the risk cats pose to wildlife and the environment (Peterson et al. 2012). These ecological risk perceptions drive tolerance for cats and perceptions of cat management (Wald and Jacobson 2013). Debate over whether or not management methods (e.g., Trap-neuter and return, euthanasia, animal shelters, etc) are practical, humane, or effective, have led to intense conflict, protests, and distrust between managers and stakeholders. Stakeholders in this issue include cat owners, feeders, animal activists, veterinarians, wildlife/bird associations, and animal shelters.

11.3.1.1 Step 1—Focus on Interests, not Positions

Take the time to find out where the parties agree and disagree. Negotiations often break down because the parties focus on areas of disagreement. This can lead to both parties becoming so entrenched in their position that the negotiation reaches an impasse. Try shifting the focus of your discussion to areas of agreement and shared goals and interests. Make sure everyone’s interests in relation to the problem are clearly identified and acknowledged. For example, competing interests in the case of outdoor cats may include concern over cat welfare, concern over wildlife welfare, frustration over the smell and noise of outdoor cats, or enjoyment of seeing an outdoor cat. Taking a step back and focusing on potential areas of mutual interests, such as desire to reduce the number of unwanted and uncared for outdoor cats, will build a sense of agreement and amity between groups. Identifying complementary interests can provide the foundation for a better agreement.

In 2007, controversy erupted over the trapping and removal of feral cats in and around the National Key Deer Refuge in Florida. Animal advocacy groups protested and public meetings were filled with “heated” discussion (Clark 2011). Instead of focusing on anger over differences in their opinions, animal welfare groups and the US Fish and Wildlife Service focused on their common goal of reducing the homeless pet population (Clark 2011). A new stakeholder group, “One Animal Family” was formed, which focused on educating the public about spaying and neutering pets and preventing free-roaming cats on or near the National Key Deer Refuge (Clark 2011). Although the problem is still being resolved, the use of negotiation principles brought all stakeholders to the table.

11.3.1.2 Step 2—Focus on the Problem, Work with the People

Building a strong and working relationship with all of the parties involved in the conflict is an important step that should take place *before* there is a conflict. Remember that individuals will come to the negotiating table with their own emotions, values, perceptions, different backgrounds, and viewpoints. Some stakeholders place a high value on cats, significantly more than birds and other wildlife, while other people value the presence of birds at their bird feeder, and other stakeholders value both. These characteristics can play an important role in the negotiation process because they contribute to blind spots, fallacies, and biases. Negotiators will have more success if they treat others involved in the conflict or debate sensitively and remember that the goal of negotiation is to reach an agreement, not win an argument. The groups should view themselves as partners in a joint search for a fair agreement advantageous to each. Key elements that will facilitate compromises are a focus on shared perceptions, clear two-sided communication, and an acknowledgement of the importance and relevance of emotions. Both sides should be encouraged to view the situation from the other parties' perspective, to assume the best of people and to acknowledge that the other side may not be directly responsible for the problem. Participants should be encouraged to talk about perceptions, emotions, values, and identity and the role they play in influencing individual perceptions. Finally, both sides must be involved in the process or neither will have a stake in the outcome.

Successful communication requires that both sides talk, listen to, and understand each other. Discussions should encourage active listening. By dealing directly with the other side and communicating that you understand their interests, you can help separate emotions and egos from the substance of the negotiation. In Portland, Oregon, negotiations over the difficult issue of feral cat management took several years before the Feral Cat Coalition, which promoted TNR, reached a meaningful compromise with the Portland Audubon Society. For taking this step to listen to and partner with a previously adversarial group, Portland Audubon's conservation director Bob Sallinger has been referred to as "the most extreme environmentalist in Portland" (Carey 2012). This exemplifies an important point about communicating at different scales (e.g., local, state, and national). The Portland effort was implemented on a small scale to address a local example of a broader issue and was popular locally, but not necessarily approved of by external groups not directly involved in the negotiation. It is also important to consider how messages or approaches may be received on a broader scale, especially when taken out of context.

11.3.1.3 Step 3—Generate a Variety of Options

At the beginning of a negotiation, both parties can feel as if their options are limited. A stakeholder dealing with too many outdoor cats in her neighborhood may feel that allowing free-roaming cats outdoors will result in wildlife deaths, while relinquishing a cat to an animal shelter will mean death for the cat. One of the best steps negotiators can take, to get away from preconceived notions or fixed ideas of what is right, is to come up with new ideas. Brainstorm first without any judgment about whether the ideas are good, bad, feasible, or infeasible. Encourage creative thinking and lots of options and save criticism of new ideas for later. At this point in the process it is important not to focus on the best or perfect solution, but a variety of possible solutions. Possible approaches to cat management include translocating colonies to areas away from key wildlife habitat, developing an active and ongoing cat adoption effort, trap-neuter and return, euthanasia, removal to an animal shelter, removal to a long-term cat shelter, providing cat collars or other deterrents to reduce cat predation, enclosing cat colonies so that cats cannot roam, keeping owned cats indoors, more responsible pet ownership, and making feeders responsible for fed colonies. Once you have your larger list of possible outcomes, it is easier to look for areas of shared interest between groups. Focusing on possible gains for both sides rather than potential losses can build rapport between stakeholders.

The groups involved in feral cat management in Portland agreed that cat predation on wildlife could be reduced by translocating feral cat colonies to areas far away from key wildlife habitat. The groups partnered to support an ordinance banning outdoor cat feeding—with the exception of a few carefully monitored and limited cat colonies. They agreed on and promoted the message: "Since pet cats allowed outside are just as likely to slaughter wildlife as their feral cousins, all felines should be spayed or neutered, kept indoors or in outdoor enclosures, and cared for. And of course, no animals should be dumped or abandoned," (Carey 2012, p. 36).

11.3.1.4 Step 4—Base the Result on Objective Criteria

It is important that the results of a negotiation are based on objective and clear criteria. The standards on which to base your agreement will vary based on the issue, but can be drawn from several sources including scientific judgment, professional standards, efficiency, cost, moral standards, equal treatment, tradition, and reciprocity. Your goals will be different whether your objective is to reduce cat predation on wildlife or reduce the current population of outdoor cats. Once objective criteria are identified, keep in mind that the proposals should be framed as a joint search for objective criteria and that compromises should be made on the basis of principle rather than pressure (e.g., bribe, threat or manipulation).

Like all communication processes, it is important to define and keep in mind your objectives to ensure you make wise decisions, including the decision to try pursuing alternative approaches. Sometimes it is helpful to bring in a mediator or third party to ensure an open discussion of interests. They can provide an impartial view and help separate the discussion of options from the actual decision-making.

11.3.2 Limitations of Negotiation

As a group activity, the negotiation process has inherent constraints and much value. Some of the criticisms of wildlife dispute resolution include the potential lack of focus on scientific information. Sometimes negotiations resolved at local levels address issues that should involve a national dialogue, or groups, such as urban environmental organizations or social welfare programs, are left out. Other critics argue that the process may include groups with unequal resources such as time, money, information, and negotiation training. Mediation can overcome some weaknesses of negotiation by giving shared control over the process to a trained mediator, who works to ensure the process is fair and trains participants to use important communication skills such as active listening. Similarly, arbitration can overcome problems associated with unwillingness to compromise by giving the third party influence over the outcome in addition to the process. It is important that negotiation processes are as transparent and inclusive as possible to ensure wise decisions are made, and stakeholders are invested in the result (O'Leary and Bingham 2003). Ensuring stakeholders understand that litigation is always an alternative to negotiated agreements is part of making the negotiation processes transparent.

Conclusion

Effective communications are essential for influencing conservation policy, changing people's behaviors, garnering funds, sharing scientific advances, and negotiating conflicts. The fate of our urban ecosystems and wildlife resources depend on effective communication with diverse audiences. This chapter described the communications process and provided guidelines for several common communication channels used by wildlife professionals in urban areas, including mass media approaches and stakeholder group negotiations. Following a systematic plan that includes identifying goals and objectives, analyzing audiences, selecting effective media and message strategies, and evaluating the impacts, should result in successful communications to promote sound wildlife management in urban areas.

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Chapter 12
Integrating Wildlife Conservation into Urban Planning

**George R. Hess, Christopher E. Moorman, Janette Thompson
and Courtney L. Larson**

Keywords Chicago Wilderness · Comprehensive plan · Development ordinance · Green infrastructure · Sonoran Desert Conservation Plan · Wildlife zoning · Zoning ordinance

12.1 Introduction and Scope

Urban wildlife conservation has the essential goal of creating, conserving, and maintaining places for species that would otherwise be displaced by people's use of the land. In this chapter, we focus on conserving habitat for native *urban avoiders*—species that typically do not persist in urban and suburban environments—at the scale of cities and metropolitan areas. These species are usually associated with the vegetative communities present prior to development of the city. Examples of displaced vegetative communities include forested habitat along the east coast of the USA, prairies in midwestern America, and desert landscapes in large parts of the American West.

G. R. Hess (✉)
Department of Forestry & Environmental Resources, North Carolina
State University, Raleigh, NC 27695-8008, USA
e-mail: george_hess@ncsu.edu

C. E. Moorman
North Carolina State University, 110 Brooks Avenue, Box 7646, Raleigh,
NC 27695-7646, USA
e-mail: chris_moorman@ncsu.edu

J. Thompson
Iowa State University, Ames, USA

C. L. Larson
Colorado State University, Fort Collins, USA

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For some wildlife species, habitat conservation applied at the local (Chap. 14) or neighborhood (Chap. 13) scale is sufficient. In the case of urban avoiders, where larger areas or the ability to move among areas of patchy habitat are required, planning and coordination are needed at the scale of landscapes covering entire cities or metropolitan areas. Acting at a larger scale can also increase the effectiveness of backyard- and neighborhood-scale activities by coordinating otherwise independent actions and creating connections—both ecological and social—that can further improve conditions for wildlife.

Before describing methods to integrate wildlife conservation into urban planning, we review the landscape-scale effects of urbanization on wildlife, key relevant principles from the fields of landscape ecology and conservation biology, and the process of land development and planning. We present two case studies highlighting different approaches to creating and managing wildlife conservation networks in urban areas before concluding with some thoughts for the future.

We devote significant attention to describing the planning process because this information is generally absent in the teaching of urban wildlife management. Comprehensive plans, zoning ordinances, and development regulations are *de rigueur* in urban America, which is the primary geographic area we cover. By determining in part *what* can be built *where*, these planning tools shape urban landscapes for wildlife in profound ways (Hair 1986). Urban wildlife conservation simply will not happen unless wildlife professionals become familiar with these concepts and involved in these processes.

12.2 Landscape-Scale Effects of Urbanization

Landscape ecology deals with heterogeneous and dynamic mixtures of different types of land cover or ecosystems, and focuses on the interactions among them. There are distinct European and North American conceptualizations of the discipline (Farina 1993). In Europe, landscape ecology emerged from the planning disciplines and therefore has well-developed ties to land use and planning (e.g., Naveh and Lieberman 1984; Zonneveld 1995; Mörtberg et al. 2007). In North America, it emerged from the field of ecology and has focused on the analysis of spatial patterns, flows of energy and materials, disturbances, and pattern-process interactions (e.g., Forman and Godron 1981, 1986; Forman 1995; Turner 2005).

For the purpose of this chapter, we can consider a landscape as the stage on which wildlife exists, comprising a mixture of different types of habitat. A *patch* is a relatively homogenous area that is different from its surroundings. Patches exist in a *matrix*, which is the dominant habitat type in the area. If there is no dominant type, a landscape may be referred to as a *mosaic*. For example, forested patches can exist in an urban matrix (predominantly urban with scattered areas of forest) or forest-urban mosaic (forest and urban in approximately equal amounts). Long, linear *corridors* of similar habitat type can connect patches, such as a riparian forest connecting two larger patches of forest in an urban matrix.

Although the presence and abundance of wildlife species in patches of habitat are influenced by the structure, composition, and distribution of habitat elements within the patch (e.g., food, nest sites), the size and shape of the patch and the matrix in which it is embedded also are important in determining the wildlife community within the patch (Donovan et al. 1997; Rodewald 2003; Shake et al. 2011). Patch size and the matrix surrounding a patch can interact as well. For example, it is likely that increasingly larger forest patches are needed to conserve urban avoiders as the matrix becomes increasingly urban.

The decline of urban avoiders in cities is caused by a series of associated mechanisms, operating at various scales, which can be categorized broadly as degrading habitat condition and disrupting connectivity. Direct loss or degradation of habitat during the urbanization process can increase competition and depredation of urban avoiders by urban adapters or exploiters, and alteration of disturbance regimes can contribute to habitat degradation. Moreover, decreasing permeability of the landscape for wildlife can disrupt connectivity among patches. The cumulative effect of these changes, if unchecked, creates a landscape unsuitable for urban avoiders.

12.2.1 Loss or Degradation of Habitat

Worldwide, urbanization has caused habitat changes leading to the loss of biodiversity (e.g., McKinney 2002, 2006, 2008). As more land is developed, less remains in native vegetative communities, fundamentally altering available habitat. Large patches of remnant native vegetation can offer a mix of elements similar to the pre-development habitat, provide interior conditions that may be different than conditions nearer to patch edges, and buffer wildlife from potentially detrimental effects of areas adjacent to patches (Donnelly and Marzluff 2004). The quality of urban habitat patches is also influenced by the amount of urban development surrounding the patch. For example, urban avoider bird species are more likely to occur in larger patches further from high density development and closer to other habitat patches or in landscapes with less urban cover (Tilghman 1987; Friesen et al. 1995; Mörtberg 2001). Similarly, the relative amount of forest and urban cover surrounding an urban wetland affects the likelihood that urban-avoiding amphibians are present in the wetland (Findlay and Houlahan 1997; Knutson et al. 1999; Price et al. 2006; Rubbo and Kiesecker 2005).

12.2.2 Increased Competition and Depredation

Habitat fragmentation from urbanization and associated anthropogenic disturbance can affect survival and reproduction of native wildlife in the urban setting (Marzluff and Ewing 2001). Urbanization may increase the density of nonnative predators and competitors. House cat (*Felis domesticus*) densities are greater near human development, and cats are a significant predator of native wildlife (Rottenborn 1999;

Sinclair et al. 2005). Similarly, densities of mesopredators, such as raccoons (*Procyon lotor*), opossums (*Didelphis virginiana*), and coyotes (*Canis latrans*), are greater in urbanized landscapes (Chaps. 7, 8). These mesopredators commonly depredate urban avoiders or their young—particularly in small, narrow patches—and may contribute to declines of these species in urban landscapes (Bolger et al. 1997; Marzluff and Ewing 2001; Schmidt 2003). European starlings (*Sturnus vulgaris*), rats (*Rattus* spp.), and other nonnative, synanthropic (human-associated) species often compete with native wildlife for food and cover in urban settings (Kerpez and Smith 1990; Bolger et al. 1997; Marzluff and Ewing 2001).

12.2.3 Altered Disturbance Regimes

Natural disturbance regimes generally are altered in urban landscapes, which can have significant long-term implications for conservation of native wildlife. Naturally occurring fires are suppressed in and adjacent to developed areas, and the ability to use prescribed fire or timber harvest to manage wildlife habitats is limited, especially where remnant patches of natural habitat are small or isolated by habitat fragmentation. Wildlife that depend on landscapes shaped by fire will be rare or absent in these human-altered landscapes unless planning and management actions are taken to conserve them. For example, early successional songbirds that rely on disturbance-created meadow and shrubland landscapes are uncommon in the mature forest patches and greenways typically conserved in North American cities (Mason et al. 2007).

People also create new disturbances that can disrupt nutrient and water cycles in urban landscapes (e.g., Marzluff and Ewing 2001; Bernhardt et al. 2008). For example, in residential and commercial landscapes where some canopy trees remain, leaf litter commonly is removed. This interrupts the normal cycling of nutrients from foliage back into the soil. In addition, the removal of litter eliminates microhabitats for many arthropods and the vertebrates that prey upon them (e.g., birds, salamanders, shrews, small snakes).

12.2.4 Decreased Permeability

Animals may need to move among habitat patches in search of food and cover, to access seasonal life history requirements such as breeding or overwintering sites, to locate available resources, to disperse and establish territories, and to maintain genetic connections with adjacent populations. Landscape permeability is a measure of how easily an animal can move through a landscape. Permeability varies with a species' dispersal ability, the degree of habitat specialization the species exhibits, and the arrangement of habitat across the landscape.

Urban avoiders typically are unable to disperse through areas of urban development, especially areas with high road and traffic densities; landscape permeability declines for these species with increasing amounts of urban land cover (Desrochers and Hannon 1997). For example, Carr and Fahrig (2001) showed the abundance of leopard frogs (*Rana pipiens*), a relatively vagile species, declined with increasing road density, possibly because of high mortality during dispersal. Further, small populations isolated in patches because of low landscape permeability may experience loss of genetic diversity through inbreeding or genetic drift, in turn making these populations less likely to adapt to environmental change through evolutionary processes (Young et al. 1996; Reed and Frankham 2003). The genetic diversity of eastern red-backed salamanders *(Plethodon cinereus*), for example, was lower for isolated urban populations than for populations in continuous habitat that allowed individuals to disperse across larger areas (Noël et al. 2007).

12.2.5 Cumulative Effect

The cumulative effect of these changes is a landscape in which urban avoiders face regional extinction, because they no longer can survive in the remaining patches, move across the landscape to meet their needs, repopulate otherwise suitable patches, or reach patches to establish new territories. Species that are poor dispersers are especially susceptible as habitat patches shrink and the landscape becomes less permeable (Bolger et al. 1997). For example, populations of pond-breeding amphibians tend to have high annual rates of local extinction and depend on repopulation from nearby ponds (Skelly et al. 1999; Green 2003). Most amphibian species are vulnerable to environmental changes that occur with urbanization (e.g., loss of forest cover, road development) and disperse poorly through the urban matrix, commonly resulting in permanent, regional extinctions of these urban avoider species in many urban wetlands (Findlay and Houlahan 1997; Gibbs and Shriver 2005).

Conversely, large patches of remnant habitat with minimal edge are the most likely refuges for urban avoiders in and around cities. Ideally, these patches would be large enough to support natural disturbance regimes (or managed disturbances that mimic them) and connected in ways that allow wildlife to move safely through the urban matrix to meet their resource needs. This is the challenge of urban wildlife conservation.

12.3 Key Principles from Conservation Biology

Conservation biology (and conservation science) focuses on the protection and restoration of Earth's biological diversity at the scales of genes, species, communities, ecosystems, landscapes, and biomes (Soule and Wilcox 1980; Soule 1986; Groom et al. 2006; Kareiva and Marvier 2012). Habitat loss, fragmentation, and degradation are among the main causes of extinction and threats to biological diversity (Wilcove et al. 1998; McKinney 2002) although there are other causes, including over-harvesting and invasive species. Because these are landscape-scale processes, there is substantial overlap among concepts from the fields of landscape ecology and conservation biology.

12.3.1 Island Biogeography Theory, Metapopulations, and Related Concepts

Margules and Pressey (2000) summarized landscape-scale conservation principles that have had profound influence on how we approach conservation (Box A). Taken together, these principles support the "node, buffer, corridor" paradigm that pervades conservation planning at the landscape scale (Harris 1984; Noss and Harris 1986; Noss 1987; Adams and Dove 1989). *Nodes* (also called *core reserves*) of high quality habitat with little or no human activity are surrounded by *multiple-use modules* (also called *buffer zones*) with levels of human activity that decline with proximity to the node; buffered *habitat corridors* connect nodes to create a habitat network. Notice the parallel to terms from landscape ecology, where nodes are patches of habitat in an urban matrix connected by corridors. Benedict and McMahon (2002, 2006) and others have adopted similar terminology when describing green infrastructure in urban contexts, referring to core reserves as *hubs* with *connectors* among them, all *buffered* from intense development by areas of low-intensity human activity.

Box A

These principles have guided thinking about defining conservation targets and the design and management of conservation networks (Margules and Pressey 2000).

Spatial autecological requirements. It is important to recognize the amount of space and different habitat types needed by species to satisfy the full needs of their life cycles.

Effects of habitat modification. As other habitat types (e.g., agriculture, urban) surround remnant natural habitat, *edge effects* can lead to changes in microclimate and species composition, altering the value of the remaining natural habitat for local wildlife. Once remnant patches shrink below a certain size in an urban setting, they become "all edge" and have no value to urban avoiders. This principle leads to "larger-is-better thinking" for patch size, as well as a desire for more circular patches to reduce edge.

Biogeographic theory. Posited by MacArthur and Wilson (1967), the theory of island biogeography states that the number of species on an island is a balance between the rate at which new species colonize the island and the rate at which species go extinct. Large islands will harbor more species and have lower extinction rates than small ones; islands closer to a mainland source will have higher colonization rates than more distant islands. These concepts lead to rules of thumb including "larger reserves are better than smaller," "closely-spaced reserves are better than distant," and "reserves linked by corridors are better than unlinked."

Metapopulation dynamics. Levins (1969) coined this term to mean a network of populations linked by dispersal. Local populations go extinct from time to time, but dispersing individuals from other populations can recolonize areas of local extinction. Metapopulations start the slide to extinction when the rate of local extinctions exceeds the rate of recolonization. Like biogeographic theory, this leads to calls for maintaining larger patches connected by corridors.

Source-sink dynamics. Reproduction exceeds mortality in source habitat; in sink habitat mortality exceeds reproduction (Pulliam 1988). Differentiating source from sink habitat is not always easy because individuals can move between them. A species can appear viable in sink habitat because of continuous immigration from nearby source habitat; this might not become apparent until connections to source habitat are severed. During the planning process, it is important to differentiate source from sink habitat and focus conservation efforts on source habitat.

Source-pool effects and successional pathways. Species composition in any place changes through time—this is called succession. Periodic disturbances in natural landscapes typically result in a distribution of successional stages, often needed for wildlife populations to persist. The distribution of successional stages is different when people suppress disturbances (e.g., fire suppression) and must be restored to conserve some wildlife species. After a disturbance, the vegetation that regenerates depends on the pool of seeds available, which, in turn, affects the animals that can use the area. This principle mandates practices that preserve the seed bank and mimic natural disturbance regimes.

12.3.2 Alpha, Beta, and Gamma Diversity

Regional approaches to conservation also have been influenced by the concepts of alpha (local), gamma (regional), and beta (spatial) diversity (Box B). Noss (1983), reflecting on measures of diversity at different scales, identified the importance of thinking regionally to conserve the widest range of biodiversity. Because many game (i.e., hunted) species are edge-adapted and wildlife management is often constrained by property boundaries, wildlife managers long focused on creating edge habitat to maximize local (alpha) game diversity for individual property owners (Leopold 1933). If everyone worked to maximize alpha diversity for game species without considering gamma diversity, edge habitat would be created everywhere to the detriment of species that require large areas of interior habitat. This would regionally homogenize species composition with consequent reduction of gamma and beta diversity. Regional knowledge, collaborative planning, and management across ownership boundaries can prevent such scenarios.

Box B

Alpha, beta, and gamma diversity measures relate to planning and management at different scales. Whittaker (1972) documented and examined three measures of biological diversity that are used commonly by ecologists.

- *Alpha diversity* is the number of species within a single habitat or small area (e.g., a backyard or neighborhood).
- *Gamma diversity* is the number of species in a large geographic region or landscape (e.g., a city or region). If all habitats in a region are similar, gamma diversity tends to be similar to alpha diversity.
- *Beta diversity* is a measure of the change in species composition across a landscape—it reflects the differences in species composition as one moves from one small area to another. Although various formulations have been proposed, beta diversity is most simply calculated as gamma diversity divided by average alpha diversity for the region. If all habitats in a region are similar, gamma diversity is similar to alpha diversity and beta diversity is low (close to 1).

12.3.3 Conservation Planning and Implementation

The essential steps for conservation planning and implementation have been presented in various forms (e.g., Adams and Dove 1989; Margules and Pressey 2000; Groves et al. 2002; Conservation Measures Partners 2007; Groom et al. 2006). In an adaptive management framework (Holling 1978), they are:

- Identify conservation targets and establish measurable goals for them.
- Map and prioritize the habitat needed to attain those goals.
- Create an implementation plan to secure needed habitat and carry out required management actions.
- Monitor and evaluate the results and adjust plans and management as needed.

In the remainder of this section, we will focus on identifying targets and mapping and prioritizing habitat.

12.3.4 Identifying Conservation Targets

Noss (2003) categorized three types of conservation targets; they are applicable in urban settings.

- *Special element* targets are rare, unique species and locations, such as remnant populations of plants, important nesting areas, large undeveloped areas, and wetlands.

Table 12.1 Urban avoiders typically have specific traits or life history requirements that make individuals, and thereby populations, vulnerable to the landscape-level change commonplace in urbanizing areas

Trait/requirement	Examples	Disruptive landscape-level change
Wide-ranging organisms	Large and medium-sized carnivores, some raptors, and some grazers	Habitat loss and fragmentation—conversion of habitat to urban uses
Species that disperse seasonally	Amphibian dispersal to and from breeding pools, turtle dispersal out of ponds to lay eggs in uplands, and northern bobwhite that disperse each spring	Decreased permeability—roads, parking lots, and buildings destroy or disrupt movement corridors—leads to increased patch-level extinctions
Species that benefit from interior habitat	Ground-nesting, forest-interior birds preyed on by urban adapters and exploiters such as domestic cats and raccoons	Habitat degradation and fragmentation—patches become smaller and more elongated—leads to increased competition and depredation
Ecosystems and associated wildlife species that require disturbances not tolerated by people	Fire-dependent ecosystems such as longleaf pine (red-cockaded woodpecker), ponderosa pine (white-headed woodpecker), chaparral (red diamond rattlesnake)	Suppression of fire to protect property
Species that require aquatic habitats for all or portions of their life cycle	Amphibians, pond turtles, freshwater mussels	Draining and filling of wetlands, loss of travel corridors among wetlands, and increased impervious surface in watersheds that leads to reduced water quality

- *Representation* targets are designed to capture ecological gradients by including areas representing the full range of biotic and abiotic conditions typically associated with an ecosystem.
- *Focal species* targets are chosen to serve as surrogates for a wider range of species and habitats (including representation targets). The notion is that conserving habitat for focal species—which usually have large home range requirements—will conserve habitat for many other species with similar needs (Lambeck 1997).

Urban avoiders are the focus of most urban conservation efforts. They typically have specific traits or life history requirements that make individuals, and thereby populations, vulnerable to landscape-level change common in urbanizing areas (Table 12.1).

These targets can be identified using data documenting occurrences of species of conservation concern, but often rely on expert opinion and local knowledge. Hess and King (2002), for example, described a Delphi survey approach in which 28 local experts selected representation and focal species targets for the Triangle Region of North Carolina through a set of structured surveys. They elicited responses

Table 12.2 Final list of landscapes and focal species for conservation planning in the Triangle Region of North Carolina. (Modified from Hess and King 2002)

Landscape	Focal species	Rationale
Extensive undisturbed habitat	Bobcat	Requires large area of habitat with relatively low levels of human activity, and preferably few roads
	Eastern box turtle	Roads disrupt connectivity between breeding habitat and other resources
Riparian and bottomland forest	Barred owl	Nests in mature, large trees and forages in bottomland. Occurs at relatively low densities
	Beaver	Keystone species that creates wetlands
Upland forest	Ovenbird	Ground nesting, interior species in mature upland forests with dense canopy
	Broad-winged hawk	Requires extensive upland forest
Mature forest	Pileated woodpecker	Requires mature forest interior habitat and breeds in large, dead trees
Pastures and grassy fields	Loggerhead shrike	Needs agrarian habitat with a mixture of open fields, scattered trees and hedgerows, forest edge, and thickets
Open and early successional forest	Northern bobwhite	Needs abandoned fields, thickets, and woodland margins. Sensitive to development

from experts, synthesized their responses, and then required them to examine and reevaluate the synthesized results. Although the process started by requesting focal species targets, the respondents suggested a combined landscape-species approach; the landscapes essentially were representation targets selected based on the needs of focal species (Table 12.2). The panel was confident that a conservation network developed around these targets, plus known locations of special element targets, would provide habitat for most species of conservation concern in the region.

In Pima County, Arizona, a Science Technical and Advisory Committee comprising scientists and local citizens developed a list of species to be used in the role of focal species targets (see case study below). They identified 55 “priority vulnerable species,” across a range of taxa, for which habitat would be mapped as a step in defining a conservation network. In the case of Pima County, this activity was part of a much larger process to develop a holistic conservation plan for the Sonoran Desert ecosystem in and around the City of Tucson, Arizona.

As these examples suggest, selecting conservation targets is a social as well as a scientific process. The citizens living in urban areas may have strong opinions about what should be conserved. These ideas often conflate biological conservation goals with recreation, aesthetics, sense of place, and other environmental objectives. It is best to recognize these differences explicitly during the planning process and attempt to formulate solutions that use them in mutually reinforcing ways (Benedict and McMahon 2002, 2006; Lafortezza et al. 2013).

12.3.5 Mapping and Prioritizing Habitat

Once conservation targets are selected, their locations must be mapped to identify potential core habitat, buffer zones, and corridors.

The locations of special element targets are sometimes cataloged, but the extent of habitat needed for their conservation may require further study and definition. In the Western Hemisphere, Natural Heritage Programs are a primary source of consistent data about known locations of species and ecosystems of special concern in geographic information system databases (NatureServe undated). Every US state, most Canadian Provinces, and many Latin American countries and states have a Natural Heritage Program. Local universities and naturalists may also have knowledge of special element targets and should be consulted.

Mapping representation targets requires explicit definitions of the gradients to be included and their identification on the ground. Typical representation targets include elevation gradients, gradients of soil and geology, and plant communities. Representation targets are often the critical habitat for focal species, as in the example for the Triangle Region of North Carolina described previously (Hess and King 2002).

Mapping the habitat for focal species targets requires defining and modeling habitat needs and finding suitable areas in the region, including core areas and corridors (e.g., Lambeck 1997; Rubino and Hess 2003). Modeling species habitat requires information about what a species needs (e.g., food, home range size, cover, affinity for edge or interior environments) and attributes of the landscape that may help satisfy those needs (e.g., elevation, geology, vegetative cover, wetlands). These data are often available in a GIS format and combined, using information about species requirements, to create potential habitat maps.

Once potential habitat has been identified, there are a variety of procedures to select the set that likely would conserve the maximum level of diversity for the least cost, in area or dollars (see Sarkar 2012 for a brief history of these algorithms). Most of these analyses rely on some combination of several basic principles (e.g., Margules and Usher 1981; Pressey et al. 1993):

- Complementarity: selecting conservation areas with minimum overlap in targets contained
- Completeness: total number of conservation targets captured
- Irreplaceability: selecting areas containing targets not contained in any other areas
- Naturalness: selecting areas that are the most natural
- Rarity: selecting areas containing the rarest elements
- Representation: capturing each conservation target at least once
- Size: selecting the largest areas of habitat

For example, one might start with the conservation hub that contains the largest number of species and select next the hub that contains the largest number of *different* species, repeatedly, until full representation is achieved. Connectors can be prioritized using algorithms derived from graph theory that allow one to determine which set of corridors provides maximum network connectivity and which connectors, if severed, will isolate large portions of the landscape (e.g., Urban and Keitt 2001; Minor and Urban 2007).

Hubs and corridors can be further prioritized for conservation based on the threat of human interference—areas most likely to be developed typically are accorded higher priority for acquisition than areas unlikely to be developed. Pressey et al. (1993), however, stressed the need for flexibility to respond to the availability of land for purchase and changing conditions. Meir et al. (2004) performed simulations that suggested strict adherence to a plan may be less effective than flexible responses to changing conditions and opportunities to protect land: sometimes it is better to protect lower priority areas when they become available rather than waiting for the highest priority areas, which might never become available.

12.4 Urban Development and Planning in the USA

Development of US cities occurs within the context of private property, is usually initiated locally by corporations and individual property owners, and is subject to a variety of interwoven planning controls established by local governments. These planning controls may be constrained by state and federal laws. People interested in wildlife conservation in and around cities must become familiar with the land development process, if they expect to influence the use of land for conservation purposes. This requires local knowledge, because the process and the cultural and political contexts in which it is embedded vary. Further, there is often a choice between fighting for conservation every time a land development project is proposed and working within the planning process to establish long-term policy that recognizes the importance of ecologically sensitive lands and creates mechanisms to avoid their development. This is a thorny issue, because excluding development typically reduces the economic value of land, which has real and significant effects on the property owners that must be considered.

12.4.1 Private Property

Most land in US urban areas and their surroundings is private property, owned by people and corporations. Within some limits, owners may use and develop their property as they see fit. These limits are defined by municipal (city) legislation that typically takes the form of comprehensive plans and zoning and development regulations. Development is often driven by the concept of *highest and best use*, which usually translates to the greatest profit that can be made by developing a parcel of land. As areas become more developed, the size of properties tends to decrease and the number of different owners increases, making it difficult to reach any kind of consensus about conservation. This phenomenon also occurs at the edges of heavily developed areas, as property owners (e.g., farmers, foresters, ranchers) anticipate development and begin splitting and selling their properties.

12.4.2 All Development is Local, Mostly

Land development is a broad term for the process of changing the landscape to meet human needs or wants, and includes everything from creating agricultural fields to city-building. Within urban and surrounding areas, development usually refers to building on lands that are currently covered by natural communities or farm fields. *Land developers,* usually referred to simply as *developers,* conceive projects and assemble parcels of land on which to build them. For example, a commercial development company might recognize an opportunity to build a large shopping center in a particular location. They will attempt to purchase property in the area and, if successful, oversee the conversion of the land to the shopping center they envisioned.

Decisions are not always solely at the discretion of developers and property owners. Some local governments attempt to control the development of cities using a variety of planning mechanisms, as authorized by citizens through local legislative bodies. These controls are codified in comprehensive plans, zoning maps, development ordinances (regulations), and transportation plans, among other documents (McElfish 2004; Duerksen et al. 2009). Where numerous cities are adjacent or close to one another, each with its own governing and planning bodies, there is often political fragmentation and insufficient coordination among them. This can complicate efforts to conserve wildlife habitat that spans multiple jurisdictions.

Local government control of land development might also be constrained by regulations at higher levels of government. For example, the US Endangered Species Act is federal law and applicable anywhere threatened and endangered species are found; local governing bodies have no choice but to work within the Act (for example, Miller et al. 2008; Stokes et al. 2010). In Washington, for example, the state's Growth Management Act established a framework, goals, and timelines that local governments must adhere to when planning for growth; decision-making is not centralized, but there are requirements and constraints within which local governments operate (Washington State Legislature 1990).

12.4.3 Overview of Urban Planning

In jurisdictions that do plan, there are numerous planning documents describing desired development within the jurisdiction and the processes through which development occurs. Land developers, who have a strong economic interest in understanding them, are well-versed in the process of creating and using these documents and are present at almost all public meetings related to them. Meetings tend to be lightly attended by others, including those with conservation interests. Advocates for the conservation of wildlife and associated habitat need to learn about these documents and processes to participate effectively (Broberg 2003; Hamin et al. 2007; Murphy and Noon 2007).

Broadly, there are *comprehensive plans* that describe the desired future for a community, *zoning ordinances* that divide the community into districts and describe the kinds of development allowed in each, and *development ordinances* that enumerate the process by which development occurs and the detailed rules and regulations for development in each zone. There may also be separate plans, integrated to various degrees, for transportation; parks, recreation, and open space; storm water control; cultural affairs; economic development; and so forth.

12.4.4 Comprehensive Planning

Comprehensive plans provide a broad perspective of the desired future for a community and a framework for arriving there (McElfish 2004; Hamin et al. 2007). They may cover a large range of issues relevant to the growth of a community, including housing, infrastructure, transportation, economic development, recreation, historic preservation, open space, and conservation (Box C). These issues are usually addressed in separate comprehensive plan *elements* that document existing and desired conditions, opportunities and obstacles to change, and recommendations and strategies for moving forward. Comprehensive plans can be sprawling documents with many inter-related, cross-referenced elements, all written in the discipline-specific language of planners. It can take significant time to understand a community's comprehensive plan and associated ordinances.

Box C

Table of Contents from the 2001 Pima County Comprehensive Plan Update, Regional Plan Policies, as Amended June 2012 (Pima County 2012a).

This document, Regional Plan Policies, is one of three working documents of the Pima County Comprehensive Plan; see also Land Use Intensity Legend and Rezoning and Special Area Plan Policies. The complete Comprehensive Plan is available in the office of the Planning Division, Pima County Development Services Department.

1. Land Use Element Regional Plan Policies
    A. Administration
    B. Cultural Heritage
    C. Site Design and Housing
    D. Public Services and Facilities
2. Circulation Element Regional Plan Policies
3. Water Resources Element Regional Plan Policies

4. Open Space Element Regional Plan Policies
5. Growth Area Element Regional Plan Policies
6. Environmental Element Regional Plan Policies
    A. Water Quality
    B. Natural Resource
7. Cost of Development Element Regional Plan Policies
8. Military Airport Regional Plan Policies
    A. Findings of the Board of Supervisors
    B. Pima County Policies
    C. Joint Land Use Study Implementation Program Plan Introduction
    D. Joint Land Use Study Implementation Strategy Policies

12.4.5 Zoning Ordinances

Implementation details are rarely included in comprehensive plans, and are found instead in zoning and development ordinances (regulations). *Zoning* is the process of dividing a community into districts and prescribing the type of development permitted in each, typically presented in a map and accompanying land use tables (Box D). Common zoning districts include commercial, industrial, manufacturing, residential of various densities, and rural agricultural. The primary purpose of zoning is to separate land uses that are thought to be incompatible and to prevent new development from harming people already using areas nearby.

Overlay zones are districts that supplement the underlying zoning with additional requirements to meet a stated goal. They are particularly useful when the goal spans multiple zoning districts, because it unifies those districts to meet that goal without the need for incorporating detailed language into each underlying zone. For example, an historic downtown overlay district might contain special provisions related to the desired character of a city’s core area.

Within a zoning district some uses are allowed *by right*, which typically means that approval is in the hands of a technical committee that reviews the project to ensure that it meets all requirements for the district. Other uses, such as those that are often considered objectionable (e.g., tattoo parlors, sex shops) or that have a potentially adverse effect on surrounding properties (e.g., large shopping centers, heavy manufacturing), require public review, hearings, and approval by the local elected body.

Box D

Development zones for Pima County, AZ (Pima County undated-a)

Rural Zones

IR—Institutional reserve
RH—Rural homestead
GR-1—Rural residential
SR—Suburban ranch
SR-2—Suburban ranch estate
SH—Suburban homestead

Residential Zones

TH—Trailer homesite
ML—Mount Lemmon
CR-1, 2, and 3—Single residence (different lot sizes)
CR-4—Mixed dwelling type
CR-5—Multiple residence
TR—Transitional zone
CMH-1 and 2—Country manufactured and mobile homes

Business

MR—Major resort
RVC—Rural village center
CB-1—Local business
CB-2—General business

Industrial

MU—Multiple use
CPI—Campus park industrial
CI-1—Light industrial/ warehouse
CI-2—General industrial
CI-3—Heavy industrial

Overlay

AE—Airport environs and facilities
BZ—Buffer overlay
GC—Golf course
H-1 and 2—Historic
HD—Hillside development

12.4.6 Development Ordinances

Development ordinances are the rules and regulations providing detailed specifications and limitations for building in each zoning district. The regulations prescribe such details as how far buildings must be from the street and adjacent property lines (setback), how tall buildings can be, how roads must be designed, and whether or not sidewalks are required. *Unified development ordinances*, which include zoning maps and development regulations, are becoming increasingly common.

If permitted by state law, development ordinances may require exactions or impact fees for the public good in exchange for project approval. *Exactions* are required mitigation of anticipated effects of a development. For example, the developer of a very large housing subdivision might be required to build a new school building or widen a roadway in or near the subdivision. *Impact fees* are direct payment to local governments for mitigation of an anticipated effect. For example, developers of smaller subdivisions might be required to pay a per-housing-unit fee into a fund that will support the building of a new school or the widening of a roadway once a certain number of new homes are built in an area.

Exactions and impact fees must be designed carefully so that they are not considered takings, which must be compensated, under the Fifth Amendment of the US Constitution. Advocates for property rights have challenged the use of impact fees and exactions, considering them a taking. For example, the Saint Johns River Water Management District in Florida demanded impact fees for wetland mitigation in exchange for a building permit. In June, 2013, the US Supreme Court ruled their demand “extortionate,” calling into question the use of impact fees and exactions nationwide (Doyle 2013).

12.4.7 Voluntary Approaches

Strong property rights sentiment exists in many parts of the US along with a philosophical opposition to zoning and development controls that force landowners to set aside land for conservation purposes. Voluntary approaches avoid these issues by attempting to develop conservation networks through the purchase of land or conservation easements from willing sellers. As of 2010, some 1700 local land conservancies in the US had taken this approach and had protected more than 190,202 km^2 (47 million acres), which is nearly twice the area of all of the National Parks in the conterminous 48 states (Land Trust Alliance 2010). Such voluntary efforts integrated into an overall, regional plan can significantly increase the capacity for conservation in urban and metropolitan areas.

12.5 Integrating Wildlife Conservation into Urban Planning

In urban areas, there is tremendous need for regional thinking, planning, and action that conserves habitat patches large enough for urban avoiders and disturbance dynamics, buffers habitat from the effects of the surrounding urban matrix, ensures that habitats are connected for life cycle requirements and metapopulation persistence, and maintains the necessary ecological processes and disturbances.

Provisions for wildlife and natural resources conservation can be included in comprehensive plans, zoning ordinances, and development ordinances. This approach is more proactive than attempting to sway decisions during development approval hearings; by then developers have already made significant expenditures on surveys and engineering and are reluctant to make substantial changes. Each step in the planning and development process provides opportunities and obstacles for people interested in wildlife conservation. In this section, we introduce opportunities for creating plans that favor wildlife conservation. There are many resources for those interested in more in-depth information (Box E).

Box E

Resources for conservation-friendly urban planning.

Environmental Law Institute (http://www.eli.org/) has numerous relevant books and research reports freely available. Their work focuses on the legal frameworks and mechanisms supporting biodiversity conservation in the US. One of the most valuable for wildlife professionals is *Conservation Thresholds for Land Use Planners* (2003), a literature review-based compendium of information about patch sizes, edge effects, corridor widths, and other design guidelines to help land use planners design viable green infrastructure.

Green Infrastructure: Linking Landscapes and Communities, by Mark Benedict and Edward McMahon (2006) is currently the definitive resource for green infrastructure development. Benedict and McMahon describe the importance of green infrastructure; key principles; the integration of ecological, social, and economic concerns; and tools available to acquire and manage green infrastructure.

In ***Natural Experiments: Ecosystem-based Management and the Environment***, Layzer (2008) presents in-depth case studies of seven attempts to conserve biological diversity in the US. Although not all are in urban settings, they all provide insights into the difficulties faced by anyone working on large-scale conservation projects. This is a must-read for wildlife professionals interested in participating in the planning or political processes; Layzer does not mince words.

Nature-Friendly Communities, by Duerksen and Snyder (2005) presents more than a dozen case studies of communities that have integrated habitat protection into their land use planning activities. They cover comprehensive plans, zoning and development ordinances, educational campaigns, and mechanisms to finance conservation.

Nature-Friendly Ordinances, by McElfish (2004) contains detailed guidance about developing comprehensive plans, zoning ordinances, and development regulations that can help local communities conserve biodiversity. Numerous examples and case studies are included. Although written primarily for planners, wildlife professionals interested in becoming active in their community planning processes will find this book accessible.

Preserving and Enhancing Communities: A Guide for Citizens, Planners, and Policymakers, by Hamin et al. (2007) is not specific to wildlife or conservation. Instead, it is a guide for people interested in participating in the various processes through which local communities govern themselves and is designed to help readers navigate those processes.

The Green Growth Toolbox (http://www.ncwildlife.org/greengrowth/), coordinated by the North Carolina Wildlife Resources Commission, is one of a growing number of technical assistance tools for communities, local governments, planners, planning-related boards, and developers. The toolbox provides mapping data, planning techniques, recommendations, and case studies for conservation of priority wildlife habitats that can be used in local land use planning, policy-making, and development design. The approach is to train interested communities in priority areas about the need for and benefits of wildlife conservation, the priority wildlife and habitats in their area, the principles of conservation biology, and how to achieve "green growth." The information is intended to provide communities with a menu of step-by-step methods and strategies to choose from to suit their needs. Recommendations center on the conservation thresholds of priority wildlife species. A model overlay district ordinance for priority natural resource conservation based on the conservation thresholds and intended for highly sensitive areas is included. (Contributed by Kacy Cook, NC Wildlife Resources Commission)

12.5.1 Wildlife-Friendly Comprehensive Planning

All US states have planning enabling statutes that *allow* biodiversity conservation to be included in comprehensive plans (Environmental Law Institute and Defenders of Wildlife 2003; McElfish 2004). Not all states, however, *require* local government to create a comprehensive plan or to include a conservation element. A comprehensive plan element for conservation might include conservation goals, provisions for developing inventories and maps that define conservation areas (e.g., core, buffer, corridor), development guidelines that respect conservation areas and allow types of development that can accommodate conservation goals (e.g., cluster or conservation subdivisions), and provisions for funding conservation.

Wildlife conservation might be included under elements for conservation, habitat, natural resources, or open space. However, information in other elements may have implications for conservation, particularly: transportation (transportation corridors fragment habitat but also provide opportunities for connectivity (e.g., greenways)), stormwater (streams and riparian areas are important conservation corridors), recreation (parks and greenways can be part of an open space network), and infrastructure (sanitary sewer lines often run along stream corridors).

McElfish (2004, pp. 34–35) suggested that a comprehensive plan's biodiversity element should (1) recognize the ecological context of the plan (e.g., ecoregion, watershed), (2) use all available data or include a requirement to collect data,

(3) provide for core habitat, connectors, and buffers, (4) address quality of life issues associated with biodiversity, (5) call for city-wide conservation measures (e.g., tree canopy, water use, native plants) that affect biodiversity, (6) address funding issues, and (7) establish accountability.

12.5.2 Wildlife-Friendly Zoning Ordinances

Because zoning ordinances control what can and cannot be developed in each zoning district, they can have a profound effect on wildlife conservation. For example, one could define conservation zones that coincide with the location of ecologically important habitat and wildlife species of conservation concern. Such actions, however, can conflict with a city's economic development goals and residents' property rights concerns. Creating a conservation zone is particularly problematic if land is already in a zone that allows development; such an action is considered *downzoning*, which is immensely unpopular with affected landowners because it diminishes the economic value of their land.

Another approach is the use of conservation overlay zones or conditional use zones that apply additional regulations and oversight to ecologically sensitive areas within the underlying zoning districts. A conservation overlay zone might, for example, span a variety of residential and commercial zones in a forested area that is home to neotropical migrant songbirds. Within the overlay zone, conservation subdivisions (Chap. 13) might be required or encouraged to retain forest overstory and leave forested corridors intact. This approach avoids the need to zone the land as open space and allows development with constraints that should not significantly reduce a developer's or property owner's return on investment.

To be effective, zoning districts intended for conservation should (1) have clearly stated conservation goals, be it for biodiversity, water quality, or wildlife habitat; (2) show clear links between conservation requirements and zoning regulations; (3) conserve as much contiguous habitat as possible; (4) include maps with defined conservation lands, buffers, and connections; (5) be supported with evidence that the zone can meet its conservation goals; and (6) be well-connected with any adjacent conservation zones (McElfish 2004, pp. 40–41).

12.5.3 Wildlife-Friendly Development Ordinances

Development ordinances detail the procedures and requirements for obtaining approval for and completing a development project, providing another opportunity to consider wildlife conservation. Development ordinances may require the developer to collect and present information about natural resources as part of the approval process. Local government can use this information to make decisions about design requirements, exactions, impact fees, mitigation, and other actions to conserve natural resources. For example, the documented presence of important wildlife habitat might trigger a requirement for a conservation subdivision on the site. Development ordinances can also include requirements for open space, recreation, or conservation land on the site of the development or in the form of an impact fee to purchase such land elsewhere for the benefit of the community.

Important legal requirements must be met when creating effective conservation-oriented development ordinances, including demonstrating a *rational nexus* (essentially a cause–effect relationship) between the development and the claimed impact, clearly stating the goals for the actions required of developers, documenting how the required exactions or impact fees are calculated based on the impact, and detailing how collected fees will be spent (McElfish 2004).

12.6 Wildlife Zoning and Green Infrastructure: A Way Forward?

In his seminal paper on ecosystem development, Eugene Odum (1969) suggested two approaches to people's use of land: compromise and compartmentalization. In a compromise approach, people maintain all land somewhere between completely production-oriented and completely natural. In a compartmentalized approach, landscape units are separated and managed for different goals, such as agriculture, cities, industry, and wilderness. Compartmentalization has long been a dominant strategy for conservation in the US (Noss and Cooperrider 1994). As has been emphasized throughout this chapter, while a compromise approach will be effective for urban adapters, urban avoiders likely need compartmentalized zones (McKinney 2002).

Here we consider how wildlife zoning may be integrated with the emerging framework of green infrastructure to address wildlife conservation in urban areas. *Wildlife zoning* organizes landscapes in terms of use by wildlife rather than people (Linnell et al. 2005). Because planners are already steeped in the nuances of zoning, using their language may improve chances for wildlife conservation—think "wildlife overlay zones." In addition, terms like *green infrastructure* (network of green space conserving natural functions for the benefit of people in cities) have potential to draw many ecologically friendly components, including wildlife conservation, into the machinery of urban infrastructure development, financing, and management (Benedict and McMahon 2002, 2006).

12.6.1 Wildlife Zoning

Wildlife zoning is the spatial delineation of wildlife conservation goals as designated zones that vary in their desired wildlife density, wildlife management techniques, or the level of protection afforded to wildlife and their habitat. Wildlife management plans typically do these things, but do not describe their management system as "zoning," a term understood by urban planners. Wildlife zoning can be used to manage wildlife across entire landscapes, from protected areas to urban centers, by integrating well-accepted zoning practices with additional zoning for wildlife that could be implemented as overlay zones.

The scale of a wildlife zoning system must correspond to the ecologically relevant scales of the species being managed (Linnell et al. 2005). Important ecological data to consider when developing a zoning strategy include species' range size, current distribution, population size and density, habitat suitability, and dispersal distance. In an urban area, information about the human population is equally important, including size and density, housing density, land use, and rates of population growth and land development.

Two special cases of wildlife zoning involve species that are hunted and species that are sometimes considered pests. *Game species zoning* is used to manage wildlife through hunting. Regulations are already common in which certain areas (zones) are opened or closed to hunters and the number and types of permits issued varies based on management goals, wildlife population sizes, and the number and density of people living in the area. *Nuisance species zoning* can be used to minimize the harm done by wildlife to people and their property (Chap. 17). In many cases, this means reducing the species' population in areas with high rates of damage.

Different zones can be created for the same species within a single city or region, depending on goals and conditions. For example, Boulder, Colorado's management plan for the black-tailed prairie dog (*Cynomys ludovicianus*) uses a nuisance species zoning approach (City of Boulder 2006). Although black-tailed prairie dogs are an important component of the prairie grassland ecosystem, they can damage landscaping and buildings, transmit infectious disease, and create roadway hazards. Following an inventory of prairie dog colonies, zones were delineated for long-term protection, interim protection (in which prairie dogs will be allowed to remain unless problems arise), and near-term removal (where prairie dogs are currently causing damage).

Wildlife zoning can be used to manage multiple species, most commonly by creating protected areas. The classic UNESCO biosphere reserve model is a form of multiple species zoning that is used worldwide. It has a well-protected core zone, a buffer zone, which accommodates limited human activity such as research and ecotourism, and a transition zone, which permits a broader range of human activity (Batisse 1982). Creating a network of protected areas linked by corridors, as described in this chapter (nodes, buffers, corridors; hubs and connectors), is an expansion of the most basic multiple species zoning strategy and is advocated by many, because it allows for increased animal movement and gene flow (Beier and Noss 1998; Hilty et al. 2006).

12.6.2 Green Infrastructure

Within the past two decades, the term *green infrastructure* has been coined and promoted when referring to interconnected open spaces that provide ecosystem services and wildlife habitat, and contribute to healthy communities and people; the approach recognizes and respects the sociological context in which conservation in cities occurs (e.g., Benedict and McMahon 2002, 2006; Lafortezza et al. 2013). The basic precepts for green infrastructure harken back to Ian McHarg's (1969) *Design with Nature*, emphasizing that development should occur after ecological conditions are evaluated and green infrastructure planned. The term is intended to resonate with planners and local government officials and put it on par with the "gray" infrastructure—such as communications, sewer, transportation, and water networks—which they are accustomed to planning, financing, and building. In the US, the concept is being applied at the scale of city (Pitsford, NY), county (Montgomery County, Maryland), metropolitan region (Chicago Wilderness), state (Maryland and Florida Greenways), and continent (Yellowstone to Yukon) (see Benedict and McMahon 2002, 2006 for these case studies).

Benedict and McMahon (2006, p. 37) stated ten principles of green infrastructure, grounded in concepts from landscape ecology, conservation biology, and urban planning. The principles emphasize well-connected, ecologically functional green space as the framework in which development should occur. Green infrastructure is described as a critical, long-term investment in a healthy community that benefits people and nature, and is created with respect for the desires of the community and individual landowners. Explicitly stated relationships among green infrastructure, ecosystem services, and human well-being are seen as crucial to the success of efforts to further develop green infrastructure in urban contexts. As with wildlife zoning, one could argue that this is simply good wildlife conservation and management repackaged, but the connections made between habitat conservation and human well-being when promoting green infrastructure are central to convincing an urban populace and governing bodies to fund its design and protection (Lafortezza et al. 2013).

12.6.3 Moving Forward

Open space in urban contexts—land that does not contain buildings and pavement (Ahern 1991) —generally occurs in parks and nature preserves that serve as core habitat for wildlife, with greenways along streams acting as corridors connecting the core habitat patches. Application of green infrastructure principles to existing open spaces in urban settings would buffer these areas from additional urban development to protect their function. Furthermore, planning for future development would occur only after core habitat and corridors for species of interest have been identified clearly (Benedict and McMahon 2006). Moreover, including such designations explicitly in comprehensive plans, zoning ordinances (including wildlife zoning), and development regulations will lead to purposeful integration of wildlife management in urban planning process and practice.

Although green infrastructure is intended to emphasize ecological function over recreational and other uses, open spaces in urban areas are often promoted as serving multiple uses that may conflict with their function as wildlife habitat, including recreation, improving human health, and transportation. Despite these potential conflicts, the multiple-use aspects of green infrastructure are important to building public support for investment in their acquisition and management. Charlotte-Mecklenburg, NC, has addressed this issue by developing parallel systems of parks and nature preserves, often adjacent to one another, but managed by distinct agencies for different uses (Mecklenburg County undated). Thus, application of wildlife zoning principles within this system could separate core habitat with little human disturbance from buffer zones with opportunities for human-wildlife interaction around the periphery of the core areas and in corridors between them. In addition to protecting core habitat for wildlife conservation, this approach encourages positive interactions between humans and wildlife along edges and corridors that will lead to important public support for additional conservation efforts (Miller 2006; Stokes et al. 2010).

As an element of this approach, greenways along streams and utility rights-of-way are an efficient conservation strategy because they are protected, provide multiple benefits (e.g., buffer water sources, provide recreational opportunities, offer aesthetic views), and often are not developable. Research in North Carolina, USA, showed that wider greenways and those surrounded by lower density development provided the greatest benefit to urban avoider songbirds (Sinclair et al. 2005; Mason et al. 2007). There are, however, potential conflicts among the various uses for which greenways are promoted. For example, Mason et al. (2007) found that forest-interior birds were more common along wide greenways with narrow trails that retained continuity of the forest canopy. Recreational users tend to prefer greenways containing wider trails with mowed, grassy shoulders because they provide more capacity for cyclists, runners, and walkers as well as longer sight lines that confer a feeling of increased safety. Greenway planners must balance competing uses carefully, if greenways are to serve conservation purposes.

12.7 Strategies for Regional Urban Wildlife Conservation: Case Studies

Here, we present case studies of two approaches to regional wildlife conservation. The Sonoran Desert Conservation Plan for Pima County, Arizona, is a holistic, landscape-scale plan with the preeminent goal of conserving biological diversity. Chicago Wilderness is a regional conservation alliance comprising some 300 organizations that coordinates efforts to study, sustain, restore, and expand remnant natural areas and engage local residents with their natural heritage in a multi-state area around Chicago, Illinois.

12.7.1 The Sonoran Desert Conservation Plan: A Biodiversity-Centered Approach

The Sonoran Desert Conservation Plan, adopted as part of its comprehensive plan by Pima County, Arizona, in 2001, was a landscape-scale approach to conserving biological diversity. It relies on a Conservation Land System that includes a set of maps and prescriptions for guiding the purchase of public protected areas and protecting biological diversity during the development process. Layzer (2008) provided a detailed account of the social and political conflicts surrounding the plan's creation, adoption, and implementation. Here, after providing brief contextual background, we focus on three aspects of the plan:

- The scientific underpinning of the plan's biological element
- The plan's insertion into the urban planning and development process
- The conditions that supported the plan's creation and implementation

Pima County, Arizona, is an arid landscape of broad plains and high mountains covering 23,800 km^2 (9190 mi^2) of land in southcentral Arizona, bordering Mexico. Approximately 305 mm (12 in.) of rain falls each year, mostly during summer monsoons. Much of the county is part of the Sonoran Desert, which has high levels of endemism and biodiversity. In 2012, Pima County was home to approximately 1 million people (US Census Bureau 2012), most living in the Tucson metropolitan area. Much of the land in Pima County is in Native American or government ownership, with approximately 14 % privately held.

12.7.1.1 Events Leading to Creation of the Sonoran Desert Conservation Plan

During the 1960s–1970s, there was little consideration of the cumulative environmental consequences of authorizing rezoning requests to accommodate construction (Layzer 2008). Relatively uncontrolled growth around Tucson sprawled into Pima County's jurisdiction, mostly toward the foothills in the northwest. New housing subdivisions often blocked access to public recreation areas; this and objections to the aesthetics of housing developments extending up the hillsides led to some of the first rumblings against this style of development (M. Livingston, personal communication). In 1972, public reaction was strong and negative when Rancho Romero, a 17,000-home, 1620-ha (4000-acre) housing subdivision, was proposed for the foothills of the Santa Catalina Mountains northwest of Tucson. Opponents, who ultimately prevailed, argued the site was better suited for conservation and recreation (Eatherly undated). Similar battles unfolded elsewhere around the edges of Tucson as environmentalists and developers became adept at blocking one another's plans (J. Fonseca, personal communication).

Several events during the 1980s let to increased natural resource protection activities. In October, 1983, Tropical Storm Octave dumped some 330 mm (13 in.) of rain in 24 h, causing major floods, significant damage of property along rivers, and visible changes to the riparian areas throughout Tucson (National Weather Service undated). In response, the Transportation and Flood Control District became active in acquiring flood-prone lands to remove buildings from the floodplain and allow for overbank flood storage and infiltration (Duerksen and Snyder 2005). Through time, focus shifted to riparian conservation linked to recreational opportunities; the river banks have become the backbone of Pima's river park and greenway system.

Pima County adopted a Hillside Development Overlay Zone (1985) to minimize damage by development and a Buffer Overlay Zone (1988) to address aesthetic concerns within one mile of designated public preserves (Duerksen and Snyder 2005). Finally, the University of Arizona's William Shaw led a small group of graduate students in creating the first comprehensive habitat map for Pima County (Shaw et al. 1986). This Critical and Sensitive Wildlife Habitats map was destined to be "just a class project" until County Supervisor Iris Dewhirst had the document adopted as a policy guideline, asking developers to voluntarily comply with its recommendations (W. Shaw, personal communication).

Despite this activity, there was no comprehensive approach to land protection, and development continued to sprawl into the desert. During the late 1990s, Pima County's citizens began electing pro-environment candidates to the Board of Supervisors; by 1998 the Board's makeup had changed from 4–1 pro-development advocates to 4–1 pro-environmental (Layzer 2008). This shift fundamentally shaped Pima's reaction to the 1997 listing by the US Fish and Wildlife Service of the cactus ferruginous pygmy owl (G*laucidium brasilianum cactorum*) as an endangered species. The owl occupied habitat in the areas north of Tucson that were under heavy development pressure. In 1998, owl sightings halted ongoing construction of a housing subdivision near the Tortolita Mountains and led to an injunction blocking construction of a high school (Layzer 2008). With development brought to a standstill, something had to be done.

County decision-makers decided to approach the challenge holistically within Section 10 of the Endangered Species Act by creating a multi-species habitat conservation plan rather than seeking a permit for the pygmy owl alone (Pima County Office of Sustainability and Conservation 2012); a Section 10 permit allows the "incidental take" of endangered species and their habitat, so long as a viable conservation plan is in place. In mid-1998, the Board of Supervisors voted in favor of this approach, noting the high financial cost of sprawl relative to more compact development and the contribution of sprawl to the destruction of the natural environment. They also adopted a Native Plant Preservation Ordinance, agreed to limit rezoning of environmentally sensitive land during the planning process (to avoid a development rush to circumvent the plan), and passed regulations allowing development rights to be transferred from sensitive lands to other areas (Layzer 2008).

12.7.1.2 Creating the Plan

The ecological centerpiece of the Sonoran Desert Conservation Plan was its Critical Habitat and Biological Corridors element. Under the leadership of County Administrator Chuck Huckelberry and Assistant County Administrator Maeveen Behan, the County recruited a Science and Technical Advisory Team, led by Bill Shaw, to develop an assessment of the region's biodiversity and a plan to conserve it. Huckelberry declared that implementation of the Board of Supervisors' directives would be based on science, that the conservation of biological diversity was the primary goal, and that the science team would be insulated from political and economic pressures and should proceed without regard to land ownership patterns and political boundaries (Layzer 2008). Public input would occur, but it would occur after creation of a plan based on conservation tenets. This action avoided a phenomenon common among stakeholder-driven biodiversity planning processes—the persistent dilution of conservation goals by political and economic considerations, such that a plan to fully conserve biodiversity is never put forth for discussion (Layzer 2008). But it also created significant tensions between County decision-makers and property-rights advocates, the real estate development industry, and other jurisdictions that they left out of that part of the planning process.

Identifying Conservation Targets The Science and Technical Advisory Team called for ensuring the long-term survival of plants and animals indigenous to Pima County through an ecosystem-based approach and established six consonant goals: recovery of federally listed and candidate species, reintroduction of extirpated species as feasible, improving conditions for species of conservation concern, reducing threats from invasive species, mitigating damage to ecosystem functions, and promoting long-term viability of species valued by people in the region (see Layzer p. 183). The team surveyed and interviewed biologists and local experts to develop a list of 55 *priority vulnerable species* to serve as focal species for habitat mapping including amphibians, birds, fish, invertebrates, mammals, plants, and reptiles (Fonseca 1999; Huckelberry 1999, 2000b; Layzer 2008). Habitats of concern were also identified for mapping and conservation action (Fonseca 1999).

Mapping and Prioritizing Habitat The team developed habitat models for each of the 55 priority vulnerable species using occurrence data (Arizona Game and Fish records plus any available state and federal records), literature accounts, and expert opinion (Huckelberry 2000a, b). All available information about the past and present distribution, life history, demography, habitat needs, and potential habitat within Pima County was compiled for each species by consulting firms under the oversight of the Science Technical and Advisory Team.

Using this information and detailed maps of vegetation, topography, geology, hydrography, and other environmental variables, the team created habitat maps for each species (Huckelberry 2000a) and combined them to create species richness maps for the County (Huckelberry 2001). Locations in which five or more vulnerable species could occur were designated as biological core areas. Areas suitable for fewer species were considered sensitive, but appropriate for multiple uses. Patches less than 405 ha (1000 acres) in size were eliminated from the reserve design, as were areas that already had been developed. "Special elements" (*sensu* Noss 1983) not otherwise captured were added to the reserve system (Huckelberry 2002). The team also developed broad areas in which connections could be made between reserves (Fig. 12.1).

12.7.1.3 Implementing the Plan

The resulting map of Biological Corridors and Critical Habitat and associated guidelines form the core of the Sonoran Desert Conservation Plan (Fig. 12.1). Important riparian areas were given highest priority with a goal of conserving 95 % as undisturbed open space; the goal for the 2125 km^2 (525,000 acres) of biological core land

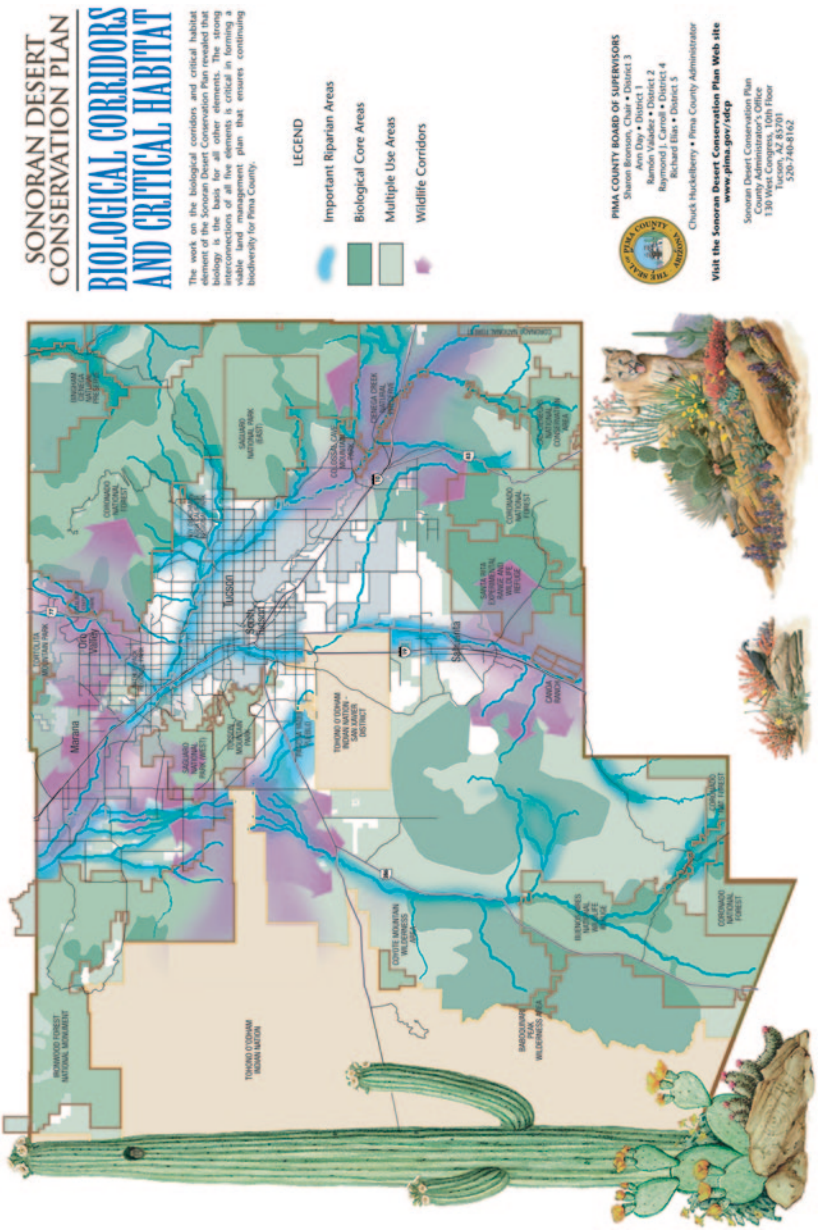

Fig. 12.1 The map of biological corridors and critical habitat for the Sonoran Desert Conservation Plan shows the biological core areas (five or more vulnerable species), the multiple use areas intended to buffer them, and areas for wildlife corridors to provide connectivity among them (Huckelberry 2000b; Pima County 2009)

was 80 % undisturbed; and for 2023 km^2 (500,000 acres) of multi-use land, 66 %. Six critical landscape connections were defined as areas where connectivity for wildlife could still be maintained or restored (Pima County 2009). The plan also included four other overlapping elements important to Pima County's citizens, providing a holistic approach to conservation: Riparian Restoration, Mountain Parks, Historical and Cultural Preservation, and Ranch Conservation (Pima County 2012b).

The County released a final reserve plan for public review in February, 2001, some two years after the Science Technical and Advisory Team began work. During that planning period, the County released numerous technical reports, held hundreds of public meetings, and assembled an open Steering Committee of more than 80 people (Layzer 2008; Davis 2009). Although they participated in presentations, the science team was insulated from political pressures and allowed to develop a plan that satisfied its mission. Once the plan was released, protests from property rights advocates, ranchers, municipalities (which had not been consulted during the process), and the real estate industry began in earnest. County leaders repeatedly discredited arguments against the plan and shifted the argument from the cost of implementing the plan (the cost had not been nailed down) to the cost of *not* implementing the plan (Layzer 2008). Despite continuous pressure from opponents of the plan, in December, 2001, the Board of Supervisors voted 4–0 to adopt the plan as part of the County's updated comprehensive plan. They also passed numerous other regulations to protect the environment from uncontrolled development (Layzer 2008, p. 192).

Land conservation occurs in several ways under the Sonoran Desert Conservation Plan's Conservation Land System. First, developers are required to demonstrate adherence to the plan's guidelines during the development approval process. Although compliance is technically voluntary, almost all development in Pima County requires a change in zoning; the Board of Supervisors is not required to agree to requests for zoning changes and rarely do if conservation conditions are not met. Second, the County raises bond funds to purchase conservation lands. The public has been supportive of bonds for open space and greenways for conservation, recreation, and economic development reasons. Third, the County has entered into agreements with ranchers in which the County purchases private ranch land and eliminates development rights on the land while allowing ranchers to continue to use the land in an environmentally sensitive manner (Layzer 2008). Fonseca and Jones (2009) reported an increase in protected land from about 182 km^2 (45,000 acres) in 2001, when the Sonoran Desert Conservation Plan was adopted, to 939 km^2 (232,000 acres) in 2009.

12.7.1.4 Keys to Success for Pima County

The Sonoran Desert Conservation Plan has long been cited as a plan with a high probability of success (Adams and Dove 1989; Duerksen and Snyder 2005; Layzer 2008). Factors contributing to this success include:

- A consistent focus on large-scale conservation efforts with biodiversity as the primary goal. Rather than focusing narrowly on obtaining an incidental take permit for the Pygmy owl, County leaders opted to create a much broader plan that galvanized public support.
- Creation of a Science Technical and Advisory Team that was insulated from political and economic pressures. This allowed the team to focus on creating for public discussion the best conservation plan rather than a plan on which everyone could agree.
- A fortuitous confluence of events and a catalyst to "force the issue." With an environmentally supportive Board of Supervisors, strong leadership from County Administrator and other staff, and a public supportive of conservation, the discovery of a federally endangered species catalyzed creation and implementation of the Sonoran Desert Conservation Plan.
- Integration of biodiversity concerns into a holistic plan covering other conservation issues important to people in Pima County: riparian areas, ranch conservation, mountain parks, and cultural heritage. For example, riparian protection is supported in part by associating it with the riverwalk and greenway system that is a popular recreational amenity.
- Leadership emerged from within the community of environmental organizations that focused their efforts. This took the form of the Coalition for Sonoran Desert Protection, which helped create a unified voice for environmental concerns. The Coalition comprises some 40 organizations that continue to support funding and implementation of the Sonoran Desert Conservation Plan.

12.7.2 Chicago Wilderness: A Regional Conservation Alliance

Chicago Wilderness is a public-private alliance among some 300 organizations (as of 2014), including civic groups, federal, state and local governmental entities, nongovernmental organizations, educational institutions, associations and clubs, faith-based groups, and corporations (L. Hutcherson, personal communication). The alliance coordinates efforts among its members to study, sustain, restore, and expand remnant natural areas and engage local residents with their natural heritage in a crescent-shaped region wrapped around the southern tip of Lake Michigan. This encompasses approximately 31,565 km^2 (7.8 million acres) extending from Milwaukee, WI, through the greater Chicago metropolitan area, eastward across the dunes and rustbelt area of northwestern Indiana, and northward into the southwestern part of Michigan's lower peninsula near Benton Harbor, MI (Fig. 12.2).

More than 2206 km^2 (545,000 acres) are under protection within this region, in parcels that range in size from 0.2 ha (0.5 acres) to 77 km^2 (19,000 acres). These lands include federal and state parks and preserves, county forest preserve and conservation districts, park districts, municipal holdings, land trusts, and other public and privately protected areas (C. Mulvaney, personal communication). The alliance serves as a collective voice for conservation efforts across a complex metropolitan region encompassing a matrix of urban, suburban, and rural landscapes. The

The Chicago Wilderness Vision

Our vision is to create a sustainable network of natural areas to provide critical homes for plant and animals, and vibrant open spaces for people.

Chicago Wilderness is an alliance of more than 300 diverse organizations protecting the lands and waters on which we all depend. We represent:
- A metro region spanning 38 counties, over 500 municipalities and more than 10 million people across parts of four states
- More than 545,000 acres of protected areas
- A potential network of 1.4 million acres of protected, restored or connected lands and waters
- More than $14 million in federal conservation funds, leveraged to $280 million over 17 years
- Over 500 conservation, sustainable development and environmental education projects funded

The Vision was created with support from the Illinois Department of Transportation and Chicago Metropolitan Agency for Planning, as well as the Gaylord and Dorothy Donnelley Foundation.

www.chicagowilderness.org

Fig. 12.2 Chicago Wilderness "Green Infrastructure Vision" showing the regional boundary, land that is already protected, and land recommended for protection. (Courtesy of Chicago Wilderness)

Chicago Wilderness alliance provides a structure for interaction and communication among a large number of participants by coordinating activities of their teams and task forces, and by hosting regional forums, a biennial "Chicago Wilderness

Congress," as well as other collaborative events that help advance goals for the alliance's major strategic initiatives.

12.7.2.1 Early History

Chicago Wilderness emerged from discussions during 1993–1996 among a group of conservation and planning professionals who had a vision of conserving biodiversity by managing the scattered natural remnants of prairies, forests, savannas and wetlands in the rapidly urbanizing Chicago region *as a single landscape* (Ross 1997). Prior to these discussions, beginning in the 1970s, there had been strong public awareness of and engagement with natural areas throughout the region, including a growing grassroots movement of volunteer stewardship and restoration (Gobster 1997). Thus, local energy and enthusiasm buoyed the early work of these partners as they developed a structure, assembled resources, and created a common vision and mission. Chicago Wilderness was launched officially at an event hosted by The Field Museum in April 1996. The buy-in and strong leadership by local and national environmental and conservation organizations provided a firm foundation for the future work of the alliance. These organizations included the Forest Preserve Districts, the Chicago Botanic Garden, The Field Museum, The Morton Arboretum, the Chicago Zoological Society/Brookfield Zoo, Openlands, the US Environmental Protection Agency, The Nature Conservancy, the US Fish and Wildlife Service, and the USDA Forest Service.

12.7.2.2 Activities

Early in the formation of what became Chicago Wilderness, participants identified five overarching goals for their work: (1) documenting and mapping the region's natural communities, (2) preventing loss of critical habitat and promoting planned development, (3) restoring natural communities on public and private land, (4) informing and engaging decision-makers and the public about the valuable natural resources in the region and the need for their management, and (5) providing opportunities for citizen involvement in conservation efforts (Ross 1997).

These goals continue to be reflected in Chicago Wilderness' work assessing and monitoring the status of natural areas and restoration activities, conducting and coordinating planning among other organizations involved in acquisition and management of natural areas (Ruliffson et al. 2002), orchestrating a strong public relations program highlighting the region's unique and valuable natural areas, and providing resources for projects and initiatives that support conservation of biodiversity in the region. They currently do so within a framework that includes a leadership group and four teams: Natural Resources Management, Science, Education, and Sustainability. The work of these teams is further organized by their efforts in support of four *Strategic Initiatives*: Climate Action, Greening Infrastructure, Leave No Child Inside, and Restoring Nature. Research is a central component of the work of the

Science team, which has recently been successful in attracting significant grants from sources such as the National Science Foundation for work combining social and ecological approaches to examining conservation and restoration decisions and activities in the region (Heneghan et al. 2012).

The alliance's approach to conservation has been strongly oriented toward identification and protection of critical areas identified as endemic, rare, or endangered plant communities (Chicago Wilderness 2011). These plant communities include 55 "subcategories" of endemic forest, savanna, shrub, prairie, wetland, cliff, lakeshore, and cultural vegetation communities (e.g., Moskovits et al. 2002). Conservation of wildlife biodiversity emanating from their work is mostly through wildlife that co-occur in these ecosystems of interest, although it is certainly made very "wildlife-friendly" by the regional framework in which this work is conducted. They have also documented wildlife diversity in some instances, including birds (Brawn and Stotz 2001), invertebrates (Panzer et al. 2010), and coyotes (Gehrt et al. 2009).

Engagement of member participants with planning expertise and representatives of local and regional planning agencies has contributed to Chicago Wilderness' effectiveness in adoption of biodiversity conservation in local and regional planning processes. For example, principles from the Green Infrastructure Vision have been integrated in the *GOTO 2040 Plan* of the Chicago Metropolitan Planning Agency, and the *2040 Comprehensive Regional Plan* of the Northwestern Indiana Regional Planning Commission (Hutcherson 2013).

12.7.2.3 Helping Create a Regional Conservation Network

The alliance has consistently conducted their work as a collaborative that effectively integrates the participation of scientists, educators, planners, policy-makers, and land managers with a regional-scale focus (Moskovits et al. 2002). Chicago Wilderness was instrumental in providing resources for the creation of an *Atlas of Biodiversity* of the Chicago Region (1997) and a *Biodiversity Recovery Plan* (Chicago Region Biodiversity Council, 1999), both geared to broad audiences. These serve as the foundation for land protection and management efforts in the region. Both documents have been updated recently, the Atlas in 2011 (Chicago Wilderness 2011) and the Biodiversity Recovery Plan Climate Change Update (Chicago Wilderness 2012), with elements added to address potential effects of climate change on biodiversity in the region.

Although Chicago Wilderness does not "own" natural areas, its work has provided strong support for the efforts of entities that do own and manage lands. Chicago Wilderness has undergirded planning, acquisition, and restoration efforts of its members by educating citizens and generating strong public support for the protection, restoration, and management of natural areas (P. Gobster, personal communication), by providing information on spatial distribution and connectivity of natural areas (e.g., Wang and Moskovits 2001), by supporting the work of restoration in those areas (e.g., Heneghan et al., 2009, 2012), and by providing information and guidance for local and regional planning agencies (Chicago Wilderness 2004;

Retzlaff 2008). In addition, Chicago Wilderness has been instrumental in development of the Metropolitan Greenspaces Alliance, a relatively new organization of several similar urban conservation coalitions, and has provided support for the Practitioner's Network for Large Landscape Conservation. Thus, Chicago Wilderness has been successful in "putting the pieces together" by identifying and mapping critical habitat, affecting acquisition and management of habitat, providing opportunities for human interactions with nature in a variety of settings, and by planning at a regional level to protect and promote biodiversity.

12.7.2.4 Keys to Success for Chicago Wilderness

Chicago Wilderness has been touted as an example of great success in urban conservation. Many factors have contributed to the alliance's ability to lead effective efforts in the region, including:

- Early and strong participation by local leaders in most cases from respected and place-based institutions including both governmental and nongovernmental organizations
- A full-court press on public relations and public education and engagement with a clear message focused on biodiversity
- Purposeful integration of biodiversity and habitat protection in planning at multiple scales (e.g., over time from Forest Preserves to multi-county and now multi-state geographic scope)
- A broad range of conservation partners
- The organization's positive approach to tackling challenges, both old (e.g., habitat degradation) and new (climate change), to biodiversity conservation in a large, heavily urban-influenced landscape

Conclusion

The integration of wildlife conservation into urban planning processes provides an opportunity to protect elements of biodiversity in the landscapes where people live, work, and play (e.g., Miller and Hobbs 2002). In the absence of specific planning to conserve wildlife, the landscape-scale effects of urbanization on wildlife populations are generally negative through degradation or elimination of habitat, increased competition with and depredation of sensitive wildlife species by generalist species, alteration of natural disturbance regimes, and decreased permeability of urbanized landscapes that inhibits the movements of urban avoider species.

Key principles from landscape ecology and conservation biology can be used to devise wildlife conservation strategies that counteract negative effects or urbanization. In an urban context such strategies must fit within well-established planning processes that include the development and application of comprehensive plans, zoning ordinances, and development regulations. These planning tools shape urban landscapes for wildlife, whether or not wildlife are considered explicitly. The emerging framework of green infrastructure provides an opportunity for wildlife conservation to become a standard element in local and regional planning efforts, and an integral part of local and regional comprehensive plans. For this to occur, wildlife professionals must become involved in these processes, either directly or through strategic alliances with people and organizations who are.

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Chapter 13
Conservation Development: Designing and Managing Residential Landscapes for Wildlife

Mark Hostetler and Sarah Reed

Keywords Conservation development · Urban wildlife · Biodiversity · Green development · Wildlife management · New urbanism

13.1 Introduction

Residential development is a leading cause of habitat loss and fragmentation, which contribute to global declines in wildlife populations and biodiversity (Sala et al. 2000; Chap. 7). In the USA, sprawl and urbanization are major sources of land use change (Brown et al. 2005), and together with introductions of nonnative species, urbanization endangers more species than any other human activity (Czech et al. 2000). Residential sprawl is a particular concern for wildlife because private lands where development most often occurs are typically lower in elevation and more biologically productive (Scott et al. 2001) and provide habitat for a disproportionate number of species (Wilcove et al. 1996; Hilty and Merenlender 2003) compared to public lands.

Approaches to protect wildlife habitat in urbanizing landscapes include the purchase of land or the purchase of development rights via conservation easements (Merenlender et al. 2004). Using these techniques, land trusts and government agencies have made important progress toward protecting wildlife habitats and other biological resources on private lands (Aldrich and Wyerman 2005). However, there remains a substantial gap in the funding needed to secure a comprehensive network of habitat conservation areas in the USA—the deficit has been estimated to be as much as $ 3–6 billion per year (Lerner et al. 2007). In addition, the rate of land development continues to be greater than the rate of land protection, and the annual

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M. Hostetler (✉)
Department of Wildlife Ecology & Conservation,
Institute of Food & Agricultural Sciences, University of Florida, 110 Newins-Ziegler Hall,
P.O. Box 110430, Gainesville, FL 32611-0430, USA
e-mail: hostetm@ufl.edu

S. Reed
Wildlife Conservation Society North America Program, 301 North Willson Avenue,
Bozeman, MT 59715, USA

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Fig 13.1 A new home built on a lot in the Madera subdivision, Gainesville, FL. Notice the medium-sized trees that have been conserved and that the landscaping contains no turfgrass and the smaller plants installed are primarily native. Green developments that are designed to conserve biodiversity must concentrate on conserving native flora and fauna both on conserved areas and built areas. (Source: Mark Hostetler)

area of land being developed exceeds that of land being protected by a factor of 10:1 (National Resource Conservation Service (NRCS) 2007). Thus, new development approaches are needed to design and manage residential landscapes for wildlife. In this chapter, we describe the challenges and opportunities associated with one possible solution: conservation development.

Conservation development is a tool that has been proposed to help fill the gaps in land protection and funding available for conservation. We define conservation development as an approach to the design, construction, and stewardship of a development property that achieves functional protection for natural resources, while also providing social and economic benefits to human communities (Fig. 13.1). Conservation developments can reduce risks from natural disasters, promote local ecosystem services (e.g., water purification), increase property values, encourage immigration and retention of highly educated residents, generate tourism income, increase quality of life, and reduce taxes through reduced infrastructure costs (Peterson et al. 2013).

There are many different types of conservation-oriented residential development projects that occur along a rural-to-urban gradient (Hostetler 2012a; Milder 2007). These range from low-density development projects in rural areas to higher density development projects in suburban and urban areas. Conservation developments commonly aim to maximize the conservation of natural areas in conjunction with developing a portion of the site. Two basic principles distinguish the conservation design process from conventional residential site design (Odell et al. 2003; Till 2001; Zimmerman 2001; Arendt 1996). First, a process of ecological site analysis is used to inventory and map the features of the site that should be protected. These features may include populations of plants or animals, terrestrial or aquatic habitats, or rare or important ecosystems. Second, a substantial portion (50–90 %) of the site is set aside as protected open space. Land protection may be achieved through a conservation easement held by a land trust or government agency, a restrictive covenant imposed by the developer, or collective ownership by a homeowner's association.

In this chapter, we explore the ability of conservation developments to conserve wildlife populations and habitats. First, we discuss the history and current knowledge about conservation development. Next, we outline strategies that improve the capacity of conservation developments to conserve wildlife species and their habitats. We end the chapter with a section on future strategies to increase the adoption of conservation developments and research priorities to understand the wildlife benefits of conservation developments.

13.2 Conservation Development

13.2.1 History of Conservation Development

Although increasing in popularity as a strategy for land conservation and sustainable development, conservation development is not new. The idea of preserving common open space within human communities traces back to the origins of urban planning and common ownership of lands in classical and medieval Europe (Carr et al. 1993). More specific ideas regarding conservation design for residential subdivisions—the ideas that private lots and protected open space should be combined in a unified design, and that they should be designed and managed to preserve their ecological values—arrived with the Garden City movement, emergence of the modern residential subdivision, and progress in landscape architecture (Weiss 1987; Parsons and Schuyler 2002). These early movements emphasized the benefits to human communities of preserving open space within residential developments, whereas later contributors added the goal of protecting natural resources.

The Garden City movement was a utopian response to the industrial living and working conditions in Victorian England. At the turn of the twentieth century, the movement advocated for the development of self-sufficient planned communities surrounded by greenbelts, in which relatively high-density residential lots were interspersed with ample open space (Howard 1902; Parsons and Schuyler 2002). At the same time in the United States, the modern residential subdivision—defined as land platted and designed specifically for residential use—appeared with the expansion of local land use planning and zoning regulations (Weiss 1987). With the associated legal, economic, and technological advancements in land development, landscape architects also recognized that integrating open space into residential developments increased their economic value and desirability (Crompton 2007). For example, Olmsted (1919, p. 14), observed that "a local park of suitable size, location and character, and of which the proper public maintenance is reasonably assured, adds more to the value of the remaining land in the residential area which it serves than the value of the land withdrawn to create it".

These ideas were echoed in the fields of wildlife ecology and conservation by Aldo Leopold's Land Ethic. Leopold recognized that conservation goals for wildlife habitat and other natural resources were not going to be achieved simply by designating large tracts of public lands. The use and stewardship of private lands—including residential areas—would also play an important role. Leopold (1949, p. 239) argued that "the land ethic simply enlarges the boundaries of the community to include soils, waters, plants, and animals, or collectively: the land."

The first guidelines for how to plan, design, and market a conservation development came in Whyte's (1964) report, *Cluster Development*. Whyte served on the Federal Outdoor Recreation Resources Review Commission (ORRRC), which recommended ensuring that Americans have access to outdoor recreation opportunities close to their homes as the nation became more urbanized. These recommendations—and specifically, the idea that protected open space within developments should be available for human use and enjoyment—were carried through to *Cluster Development*. The report evaluated the challenges and opportunities associated with implementing conservation development, but generally advocated for widespread adoption of conservation design techniques in residential development.

In his next book, *The Last Landscape,* Whyte further argued that "open space planning should take its cue from the patterns of nature itself—the water table, the floodplains, the ridges, the woods, and above all, the streams" (Whyte 1968). In essence, this was a call for ecological site analysis. Methods for how to plan a development with consideration of the natural landscape appeared in McHarg's (1969) *Design with Nature*. A prominent landscape architect and planner, McHarg provided guidelines for 'ecological design,' or an approach that first inventories the physical and biological features of a site and incorporates them into the design of the built environment.

The peaks of interest in conservation design in the USA can be interpreted as responses to waves of expansion of urban, suburban, and exurban development at key points in the country's history. The modern residential subdivision, with landscape architecture's contribution of common open space, emerged following World War I and a series of federal subsidies to stimulate road and land development in the 1920s (Hayden 2004). Calls for cluster development and an ecologically informed site design process were alternatives to the sprawling post-World War II suburban residential subdivisions, from Levittown, New York, to the San Fernando Valley in California (Jackson 1985).

In the 1990s, economic prosperity and large-lot developments at the rural fringes of metropolitan areas combined to consume more land per capita than any other type of development in the United States (Brown et al. 2005). This most recent phase of rural sprawl set the stage for contemporary interest in conservation development. The best-known modern advocate for conservation design is Randall Arendt. In *Conservation Design for Subdivisions*, Arendt (1996) offered examples and guidelines for how to implement a conservation development. He also argued that the key to widespread implementation of conservation development is to ensure that enabling conditions are present in local land use codes. Specifically, he suggested that subdivision and zoning regulations must permit clustering and offer guidelines for ecological site analysis and conservation design techniques.

13.2.2 Current Knowledge About Conservation Development

Conservation developments are increasingly common, especially in rural areas of the USA, and interest in the planning and design technique is growing among land use planners, developers, and conservation organizations. Land use regulations that establish guidelines or create incentives for conservation development have been adopted by nearly one third (32 %) of local jurisdictions in the western United States and one fifth (19 %) of local jurisdictions in the northeast (Reed et al., unpublished data). The rate of adoption is increasing rapidly, and the majority of conservation development ordinances have been adopted within the past 10 years. Nationwide, 2.5 % of residential subdivisions are estimated to be conservation development projects (McMahon and Pawlukiewicz 2002), equivalent to more than 40,000 new housing units per year. The proportion may be greater in rural areas, particularly those near urban centers with highly educated residents (Allen et al. 2012, 2013). For example, our colleagues found that up to 16 % of new housing units in unincorporated areas of Colorado counties from 2000 to 2010 were located in conservation development projects (Mockrin et al., Rocky Mountain Research Station, USDA Forest Service, unpublished data). Conservation developments are also making a substantial contribution to conserving natural or seminatural open space on private lands. Defined broadly to include conservation easements with reserved home sites, a recent study estimated that conservation developments account for up to 25 % of the private land area protected in the United States (Milder and Clark 2011).

The dual goals of conservation developments are to protect natural resources and to generate benefits for human communities. Definitions for conservation developments vary not only in terms of housing densities and amount of open space, but also in terms of the primary stakeholders involved with the project and terminology used to describe the development. For example, land trusts typically initiate conservation and limited development projects (CLDPs) and for-profit developers typically initiate conservation subdivision designs (CSDs) (Milder et al. 2008). CLDPs typically allow only 5–25 % of permitted density and proceeds are used to finance the management of the remaining conserved parcels, whereas CSDs are typically developed at 100–200 % of permitted density to maximize profits and proceeds may or may not go towards conservation activities. Despite the strong potential for conservation development to protect wildlife habitat and to generate revenue for development infrastructure and ongoing management, species conservation is rarely a major focus of the technique. For example, in most texts important in the evolution of conservation developments—with the notable exception of McHarg (1969)—wildlife species are rarely mentioned as conservation targets, and specific guidelines for how best to protect their habitats are missing (Hostetler and Drake 2009). This is partially due to the fact that conservation developments are designed to achieve a wide array of other goals, including agricultural, cultural, recreational, and aesthetic goals (McMahon 2010). It may also be attributable to the fact that conservation design and management standards have been developed principally by land use planners and designers, with limited involvement of wildlife ecologists and other scientists.

Much like conservation easements (Merenlender et al. 2004), conservation developments are rarely evaluated for their effectiveness at protecting their conservation targets or other nontarget natural resources over the long term. From the few empirical studies, there is variation in conservation benefits among the different types of conservation development projects. In northern Wisconsin, mapping and modeling analyses demonstrate that conservation development reduces the rate at which habitat is lost due to housing growth (Gonzalez-Abraham et al. 2007), and in Falmouth, Maine modeling analyses determined that conservation development policy could generate a more permeable, interconnected network of open space (Freeman and Bell 2011). In the eastern United States, CLDP projects were more effective than CSD projects at protecting and managing threatened natural resources (Milder et al. 2008). In theory, clustering should reduce the impacts of residential development on the composition and abundance of native wildlife, because the zones of influence around individual homes would overlap (Odell and Knight 2001; Odell et al. 2003; Theobald et al. 1997). However, homes in conservation developments are often clustered near sensitive resources (e.g., lakes) and the developed areas can impact the conserved areas (Gonzalez-Abraham et al. 2007; Hostetler 2012a). In a recent field study, clustered and dispersed housing developments did not differ in the composition of songbirds, mammals, or nonnative plants (Lenth et al. 2006). The authors speculate that the lack of difference between conservation and conventional developments may be due to the relatively small size of the conservation development projects, off-site effects of human disturbances on adjacent properties, or human use of protected open space within the conservation development projects.

Overall, conserving open space is an important step, but cluster designs and conservation of critical natural areas alone do not guarantee protection of wildlife species and habitat because nearby developed areas can have a variety of negative impacts. It is critical that wildlife ecologists participate in evaluating conservation developments for their effectiveness at meeting their conservation targets. Conservation development projects will continue to yield economic and social benefits for developers and communities (Mohamed 2006; Bowman et al. 2009), regardless of whether they are making a meaningful contribution to protection of wildlife species and other natural resources. In the next section, we provide suggestions on how conservation developments should be designed and managed to benefit wildlife populations.

13.3 Designing and Managing Functional Conservation Developments

The architecture community is beginning to include ecological principles in the design of buildings and subdivisions (Arendt 2004; GhaffarianHoseini 2012; Alwaer and Clements-Croome 2010; Anselm 2006). As part of this movement, conservation developments have the potential to conserve wildlife species and wildlife habitat, but even with the best intentions, the original intent of a community can fail because of the complex interactions among the three phases of building and managing residential communities. These are the design, construction, and postconstruction phases (Hostetler 2012a). During the design phase developers (collaborating with environmental consultants) draw out lot lines, designate open space, and place roads throughout the site. At this stage, goals for the project are discussed and prioritized. If a CSD is preferred, homes would be clustered in a defined space and the remaining area designated as open space. Basically, everything is laid out on paper and it is more about where things go and the juxtaposition of vertical (e.g., buildings) and horizontal structures (e.g., roads, lots, conserved areas).

Next, during the construction phase, contractors and subcontractors implement the plan, constructing homes, streets, waste treatment systems, and landscaped areas on individual lots and common areas. During construction, earthwork machines can compact soil, destroy understory vegetation, and introduce invasive nonnative plants that may spread into any conserved wildlife habitat. Even the improper maintenance of silt fences can lead to stormwater pollutants (e.g., silt and nitrate) entering into wetlands and destroying the habitat.

Postconstruction is the last phase, where buyers purchase the homes, move into a community, and manage their homes, yards, neighborhoods, and common areas (Chap. 14). Collectively, individual decisions have dramatic impacts on both built and conserved areas in terms of biodiversity conservation (Hostetler 2012a, Hostetler and Drake 2009).

Although all development projects go through these three phases, the construction and postconstruction phases often receive less attention from planners and conservationists. Even the best designs will fail to achieve wildlife conservation goals unless they are implemented in the construction and postconstruction phases (Hostetler 2010, 2012a; Hostetler et al. 2011). Long-term management of neighborhoods is the last link in the chain determining how wildlife-friendly a neighborhood ultimately is. Below, we highlight wildlife conservation strategies for the design, construction, and postconstruction phases. Under each, we offer case studies, strategies, and practices that will most likely benefit wildlife populations and help conserve or restore biodiversity within a proposed conservation development.

13.3.1 Design Phase

During the design phase, potential impacts on wildlife and biodiversity must be considered alongside other priorities including marketability, development costs, and transportation. Many connections exist between biodiversity conservation and other natural resources, livability, and economic issues; a good design will incorporate multiple values and almost any site can incorporate biodiversity conservation alongside other priorities. However, each site is unique and a biodiversity rubric or algorithm cannot be consistently applied across sites. We outline some considerations below that will help one to create a good conservation design.

13.3.1.1 Inventory

First, one must conduct a thorough inventory of wildlife populations and habitat that can support wildlife through different stages of their lives—including areas for breeding and rearing offspring, areas that promote movement from their birthplace (including stopover sites for long distance migrating species), and areas for overwintering. A good inventory requires evaluation of how a proposed development site can accommodate animals that reside on a site year round and those that may only use a site during certain times of a year. As an example, many bird species in North America breed in the northern latitudes during the summer and then migrate south during the fall/winter and patches of habitat along their migratory route could be used as a stopover site where birds forage or seek cover. Thus, an urban conservation area and even small landscape areas could serve as a breeding site, a stopover site, and/or a wintering site (Hostetler et al. 2005; Hostetler and Holling 2000).

Second, vegetation surveys are required to determine areas that have the least amount of human disturbance and contain the greatest diversity of native plants. Areas with a history of minimal human disturbance, at least in the near past, usually contain a diversity of native plant species and have the greatest potential for wildlife habitat. These areas should be designated as priorities and conserved where possible.

13.3.1.2 Connectivity

A development site should be evaluated in the context of surrounding environments. Opportunities could exist to connect wildlife habitats that occur outside of a development's boundary. Planning for wildlife corridors helps to promote the movement of species throughout large regions. A body of literature surrounding metapopulation theory suggests healthy regional populations are dependent on the occasional migrations of individuals from one subpopulation to the next (Hanski 1999). This theory predicts that wildlife populations are less likely to go extinct due to stochastic events (such as disease, tornados, hurricanes, etc.) when connected to other populations. Connectivity in the landscape allows individuals to move from one area to another, hedging against the possibility of catastrophic events causing extirpation of local populations (Keymer et al. 2000).

Roads and aggregations of buildings are the primary culprits that affect wildlife movement (Chap. 15). Ideally, roads and buildings can be constructed in areas that minimize barriers to wildlife movement, but roads and buildings often bisect natural areas. In these cases, culverts can be used to promote the passage of animals under roadways; below, we highlight some important design and management procedures to minimize roadkill and promote the safe passage of animals.

For terrestrial animals, the placement of culverts should be selected by a trained wildlife biologist. If sited and constructed properly, a variety of animals will use these underground passages (Ruediger 2001; 1000 Friends of Florida 2008). Some important principles include:

1. Dry areas must be present in a culvert for terrestrial animals, even during extremely wet conditions.
2. Culverts with natural substrates, such as soil and vegetation, work better than those using steel or other human-made material.
3. Vegetation should be present around the culverts to provide shelter and cover for animals.
4. Bright lighting should not be placed in these areas because many animals avoid well-lit areas.
5. Fencing may be required to funnel wildlife into culverts. Alternatively, the road can be raised with cement lining the sides; this prevents animals from climbing onto the road and funnels them through the culverts. Here, vegetation growing over the sides of the roads must be cut back so it does not allow wildlife to climb onto the road.
6. Skylight grates on top of the road will help with lighting the interior of wildlife passages under roads.
7. Where culvert crossings occur, signs should be posted and speed bumps installed to slow traffic because some wildlife will find ways to circumvent the culvert.

Fish and other aquatic animals, are more apt to use a culvert that is designed to mimic natural waterways (U.S. Department of Transportation, Federal Highway Administration 2007; Ruediger 2001). Important design principles for aquatic wildlife crossings include:

1. At least 10 cm-deep water is necessary to promote the movement of most native freshwater fish.
2. Avoid barriers to fish movement including fast-flowing or polluted water going through the culvert, flap gates, and overhanging culverts.
3. Include natural substrate and rocks that slow water flow and provide resting habitat within the passageway.

13.3.1.3 Cluster Design

Clustering results in greater home densities, leaving more opportunities to conserve large patches of wildlife habitat. In cases where good wildlife habitat may be scattered across a site, we do not recommend that built and conserved areas be scattered across a site. Fragmentation, by definition, creates more edge habitat and a variety of wildlife species are negatively affected by edge. For instance, many interior forest birds avoid edges and do not use small patches that are primarily edge (Bock et al. 1999; Maestas et al. 2003; Brand and George 2001). Plus, a good development design should explore land uses just outside the boundaries of a development. Opportunities exist to create larger patches of wildlife habitat when conserved space is located near natural areas just across the boundary. Several native species of birds, for example, were found more often in residential developments situated near urban parks in Phoenix, Arizona (Hostetler and Knowles-Yanez 2003; also see Chaps. 7, 12). Thus, judicious placement of designated conserved area can leverage and improve habitat in adjacent protected areas.

In addition, a cluster design can save a good deal of money and energy and reduce greenhouse gas emissions. In a case study on a proposed Florida development (called Restoration), significant savings were predicted when a conventional design was compared to a compact design. This large development had 8500 housing units; assuming each unit had one vehicle, internal and external vehicle miles traveled (VMT) were reduced from 955,726 km to 562,449 km for all the units in the subdivision (a reduction of 41 %). This translated into a reduction in CO_2 emission of 41 %, amounting to a reduction of over 4500 kg CO_2e/home/year (Program for Resource Efficient Communities, University of Florida, unpublished data). Reduced road construction associated with the compact design saved $ 150,000,000.

13.3.1.4 Biodiversity and Wetland Conservation

A fragmented landscape is not hospitable for a variety of sensitive plant and animal species due to increased edge effects of small, isolated remnants of habitat (e.g., Paton 1994). Small, isolated wetlands surrounded by built areas can be impacted by daily events such as stormwater runoff from impervious surfaces. In some situations these wetlands may actually become ecological sinks. Ecological sinks are places that attract wildlife with resources such as food or water, but ultimately harm populations because resident wildlife die at high rates. In the case of small wetlands surrounded by development, pollution and automobile collisions can rapidly kill wildlife attracted to the area. In these instances, it may be better to conserve larger healthier wetlands and actually fill in the small wetlands or use them as enhanced stormwater retention areas. The point here is that careful consideration must be given when evaluating wetlands for their conservation potential. Often, this is not considered during the design phase of a project and regulators and developers must communicate clearly their objectives. This strategy runs the risk of being abused by developers that may wish to fill in wetlands for other reasons beyond biodiversity conservation, but if biodiversity objectives are clearly stated and the company has a good conservation track record, then compact communities could have a design that incorporates conservation and creation of "good" wetlands and building on other more degraded wetlands.

13.3.1.5 Are Small, Isolated Patches OK?

The above suggestions assume that opportunities exist to conserve large patches and create wildlife corridors (Chap. 12). However, many development situations do not present these opportunities and developers may wonder whether it is important to conserve small, isolated, fragments. We say emphatically that these small fragments are important as well. Many wildlife species, including insects, are small and have small home ranges, and even one native tree can offer significant food and cover. For some species, a scattering of trees in a neighborhood can be an important food and cover resource during migration and wintering seasons. Many bird species will forage within small tree patches across a city, that could also serve as stopover habitat (Dawson and Hostetler 2010; Hostetler et al. 2005).

13.3.2 Construction Phase

Even if a design maximizes wildlife habitat conservation, the habitat can lose its integrity if the construction phase is not managed properly. The actions of contractors and other builders could destroy areas set aside for conservation in the design phase. Below, we describe how construction can be managed to promote conservation goals established in the design phase.

13.3.2.1 Trees and Natural Areas

Ironically, construction is destructive by nature. Construction activity must be closely regulated because the use of heavy earthwork machines and site grading can destroy key ecoststtem functions in a planned development.

Trees and their root systems must be protected from damage during the construction process. Otherwise, heavy machinery will damage roots, and eventually kill trees (Ruppert et al. 2005). Fencing can be used to keep heavy vehicles away from both tree trunks and the root zone where soil compaction can kill trees. It is not sufficient to fence or flag the trunks of trees. The roots underneath the drip line (the outer edge of the leafy canopy) also should be protected by a sturdy fence. Even this precaution leaves 50 % or more of the roots impacted by construction. Over 90 % of soil compaction occurs within the first three passes by machinery (Soehne 1958), and vehicle activity can cause soil to become so compact that little water can percolate into the ground (Gregory et al. 2006). Another factor that will help ensure the survival of a protected tree is irrigation. Stressed trees need plenty of water during the construction process; stressed trees should be irrigated so that water reaches a soil depth of 30 cm about 2–3 times per week. A 10–15 cm layer of organic mulch within the tree protection zone also helps maintain soil moisture. Further, lowering or raising the soil grade around trees (particularly in the tree protection zone) even a few centimeters can effectively kill a tree. Lowering the grade removes important root mass and raising the grade with fill dirt smothers the roots and prevent oxygen from reaching them. Other impacts can involve the placement of utilities. Typically, most utility lines (cable, telephone, etc.) are laid by digging trenches throughout the site. If the site is heavily forested, trenching can impact a large number of roots. The best solution (and most cost effective) is to lay the utilities beneath the roads in the subdivision.

To best conserve important natural areas within the development boundary, management of construction activities is critical. Silt fences must be well maintained around any wetlands or water bodies to prevent silt from entering these areas during construction. Runoff can carry vast amounts of silt into a wetland reducing penetration of light and dissolved oxygen necessary, while increasing nutrients, algal blooms and nonnative plants (U.S. Environmental Protection Agency (USEPA) Office of Water 2000; Bilottaa and Brazier 2008). Thus, education of contractors about the benefits of fencing is important. Earthwork machines parked near natural areas compact soil, effectively decreasing the ability of buffer areas to remove sediment and pollutants from runoff. Therefore, transitional buffer zones that are kept free of vehicle traffic are needed to minimize the effects of nearby construction activity (Fig. 13.2).

Invasive plants thrive in disturbed postconstruction landscapes, and may be introduced from off-site by construction machinery. Thus, earthwork machines should be monitored for invasive plant material throughout the construction process. To further reduce the impacts of earthwork machines and extensive grading, additional tips are offered below (Thompson and Sorvig 2011; Hostetler 2012a):

1. Use stem wall or other construction techniques that raise the floor of houses instead of using slab on grade. Often, fill dirt is required to raise the grade of the lot to meet flood requirements and the home's foundations are placed on top of the fill dirt. However, if one uses stem wall construction, only the footprint of the home is raised up the required amount to meet flooding standards. This way, the whole site does not need to be graded; the topsoil is conserved on a lot-by-lot basis, retaining important soil biota and organics.
2. Designate and clearly mark construction site access and routes that coincide with eventual streets and roads. This will limit compaction of the soil to areas that contain roadways for the subdivision.
3. Designate parking and stockpiling sites for vehicles and building materials. One should limit these areas and clearly mark them so contractors know where to park vehicles and to mix or store materials. Equipment and building materials should be located in areas that are planned for future hardscapes, such as patios

Fig 13.2 Earthwork machines can have large impacts on open spaces conserved for biodiversity. In Florida, contractors typically park their vehicles in forested areas to escape the heat. Running or parking earthwork machines under trees has the unintended consequence of killing the trees and even compacting the soil to the point where it reduces percolation rates of stormwater running off of impervious surfaces. (Source: Jeanette Hostetler)

and pavement. This technique will reduce soil compaction and contamination that could cause the demise of nearby vegetation.

4. Develop environmental covenants and contracts for contractors and subcontractors. In particular, contracts should clearly identify areas and landscape features that are protected; financial penalties should be listed for contractors that damage these areas. Bonuses and other rewards could be included for contractors who do no damage to protected areas.

13.3.2.2 Stormwater Treatment Practices for Water and Wildlife

A stormwater treatment train is a series of components linked together from the top to bottom of an urban catchment, and is designed to control water quantity and improve water quality. Recent advances in low impact development (LID) have incorporated treatment features that lengthen, slow, and filter the passage of water moving through an urban catchment. Through bioretention and infiltration, biological, chemical, and physical properties of plants and soils act to filter and improve the quality of water across a landscape. An LID infiltration system includes a variety of bioretention areas (e.g., green roofs, permeable pavements, swales, and rain gardens). If a natural wetland or existing water body is used as the end of a stormwater treatment train, one must implement a design and management system that minimizes the influx of nutrients and pollutants. Even with a good LID system, a natural wetland could fill up with sediment fairly quickly, become polluted, or become overrun with nonnative plants. The end result would be a loss of native flora and fauna within the wetland, and even loss of the wetland itself.

An LID stormwater treatment train can benefit wildlife and simultaneously address water quantity/quality issues. The use of native plants in swales and rain gardens helps promote plant diversity and ultimately wildlife diversity. Where retention or detention ponds are still needed at the end of a treatment train, these can be designed to benefit wildlife, particularly bird and fish species. The shoreline edges of retention ponds should include open areas and areas with native tall and shrubby vegetation. The area within the first few meters of a pond, or the littoral zone, should contain a mixture of open water, floating vegetation, and both short and tall emergent vegetation. A mixture of vegetation structures in both zones will attract a wide variety of wading birds. To have a food-rich littoral zone though, the edges of a stormwater pond must not be dug too deep. Instead, a gently graded shoreline will provide shallows for dabbling and foraging birds. The combination of a diverse shoreline and littoral zone attracts a diversity of wildlife species (Traut and Hostetler 2003, 2004; White and Main 2005). Good stormwater design needs to be supplemented by strict fertilization, irrigation, and yard waste management practices for yards that drain into the stormwater ponds. Nutrients from fertilizers and yard clipping can quickly cause a pond to become eutrophic and surface weeds and algal growth can take over ponds.

13.3.2.3 Lighting and Landscaping Pallette for Wildlife

Bright lights threaten wildlife by disrupting biological rhythms and interfering with the behavior of nocturnal animals (Chap. 9). For example, artificial lights can alter frog foraging and calling behaviors and some bird species avoid nesting or roosting near bright lights; newly hatched sea turtles are lured away from the ocean by artificial lighting; and migrating seabirds and other birds can be lured by lights into cities, where they can become lost and disoriented (Rich and Longcore 2005). For lighting along roads, we recommend full cut-off illumination that shields light from being emitted upwards toward the night sky or surrounding natural areas. To be fully cut-off, the light bulb should not extend below a lamp shade. The idea behind such full cut-off illumination is to both reduce night pollution so that people can view the night sky and to reduce illumination of nearby wildlife habitat that could disrupt natural wildlife behaviors. Where key wildlife species or wildlife areas are identified, one should direct road or home lighting away from critical wildlife habitat or bird flight paths. For more information on international dark-sky efforts, see DarkSky (http://www.darksky.org/).

A landscaping palette for individual lots and common areas must include native plants to benefit wildlife (Chap. 14). Using native plants not only benefits biodiversity by increasing the number of native and endemic plants in an area, but using native plants increases the amount of food and cover for wildlife. Bird diversity increases with the presence of native plants (MacGregor-Fors 2008), native plants serve as host plants for butterflies (Collinge et al. 2003), and native bee diversity increases with the abundance of native plants in cities (McIntyre and Hostetler 2001). Additionally, reducing the amount of manicured lawns benefits wildlife because a monoculture of nonnative turfgrass is no better than concrete for most wildlife species, offering little cover or food. Even the conservation of dead trees (i.e., snags) should be a key priority during construction because a variety of wildlife species use snags for nesting and foraging habitat. Few tree snags remain in urban areas but such structure is important habitat for a variety of cavity-nesting birds (Blewett and Marzluff 2005). When lots are sold to builders, developers can encourage or require builders to landscape with natives by having covenants and landscaping regulations that require natives. Landscape and construction plans should be submitted to a landscape review committee composed of knowledgeable landscape architects and wildlife biologists.

13.3.2.4 Post-Construction Phase

Although good design and good construction practices may have been implemented, residents ultimately take over management of homes, yards, and neighborhoods and determine impacts on wildlife species and habitat. Recent evidence suggests that homeowners, even in “billed” conservation developments, are not environmental stewards and do not understand the function of natural areas for wildlife. In two Florida studies, homeowners in conservation developments and conventional developments showed no difference in knowledge, attitudes, and behaviors (Youngentob and Hostetler 2005; Hostetler and Noiseux 2010). The biological integrity of conserved areas within a development can be compromised by the daily activities and decisions of many homeowners; in addition, lands just outside of the development boundary can be heavily impacted by activities originating from built areas (Meurk and McMurtie 2006; Hostetler 2012a). Uninformed homeowners can negatively impact wildlife populations a number of different ways, which include promoting the spread of invasive plants and animals, excessive pollutants in stormwater runoff, replacing native vegetation with nonnative species, over use of pesticides, allowing pets to invade conserved areas, and all-terrain vehicles (ATVs) overrunning natural areas (Hostetler 2012a). In a study on conservation developments in Colorado, researchers reported that conserved open space was dominated by nonnative plants and researchers speculated this was due to nearby residents not understanding the importance of maintaining native species on their lots and preventing the spread of nonnatives into the conserved areas (Lenth et al. 2006). Development policies typically do not address the long-term management of built and conserved areas. Very few open space policies in Florida or North Carolina addressed long-term management of built and conserved areas (Wald and Hostetler 2010; Allen et al.

Fig 13.3 Engaging local residents about biodiversity conservation is important in order to retain the biological integrity of the entire site. Here, an interpretive sign was installed along a public trail; the information on the sign discusses ways to landscape a yard to provide habitat for wildlife. (Source: Mark Hostetler)

2012). Lack of engaged residents and proper long-term management plans for built and conserved areas forms a recipe for failed wildlife conservation.

For conservation developments to be successful, long-term management is key, and it is critical that residents are engaged and understand how their actions can affect the biological integrity of the entire development. Below, we suggest some strategies to inform and engage residents and to put in place long-term management plans for built and conserved areas.

1. A developer can install a neighborhood education program in a conservation development. This program should be highly visible and educational content should be easily updated. One study found that a neighborhood educational program, which consisted of a brochure, Web site, and educational signs along trails/sidewalks, had improved awareness and confidence of residents to implement conservation practices (Hostetler et al. 2008; Fig. 13.3). Content for these educational signs were switched out quarterly to present new issues to the community.
2. Covenants, conditions, and restrictions (CCRs) are limitations and rules that govern master planned communities and a home owner association is the governing body that enforces these rules. Established by the developer, CCRs provide a legal framework that sets the "tone" for the community and provides legal recourse for aberrant behaviors. A CCR document should contain language that addresses the management of yards and conserved areas (Hostetler 2006).

3. Long-term management plans will only work if there is a dedicated funding source for implementing management practices. To secure permanent funding, a portion of lot sales could be required to go into a management fund; a portion of homeowner association dues could be allocated towards management; and where development rights are sold (e.g., conservation easements), a portion of the proceeds are retained for future management actions.

13.4 Future Directions

In this chapter, we have argued that while conservation developments have great potential, a good design must be complemented with thorough attention to construction and postconstruction phases, including guidelines for long-term management of built and conserved areas. To benefit wildlife, a variety of stakeholders must be engaged and understand the intent of a conservation development. Two key questions must be answered to increase the implementation of *functional*, conservation developments—(1) What are the mechanisms and strategies to change the current path of conventional development? (2) What are the research priorities to increase understanding of how conservation development benefits wildlife?

13.4.1 Enabling Conditions for Conservation Development

Planners and policy makers can create the enabling conditions for developers, builders and citizens to implement design and management strategies that aid local wildlife populations. This requires a unique combination of voluntary and regulatory policies, combined with education. While regulatory policies can require people to adopt new practices, often this is met with resistance form stakeholders and very few regulations stand the test of time. On the other hand, incentive-based policies provide economic benefit in terms of tax breaks, fast-tracking, and density bonuses rather than mandating practices (Romero and Hostetler 2007; Kihslinger and McElfish 2009). It was found that successful incentive-based policies, where adoption rates were high, had the following three ingredients (Romero and Hostetler 2007):

1. Stakeholder design: Incentive-based policies had a higher rate of adoption when they were formed with input from key stakeholder groups. In failed instances, incentive-based policies were formed solely by a few enterprising individuals within a county/city government—only to find out once it was released, the incentives were not really incentives to the people the policy was intended to affect.
2. Marketing campaigns: Successful policies were marketed throughout a community and people were aware of the financial and environmental benefits. Even well-crafted policies sat on the books because very few people from the outside

were aware of them. Ones that were successful usually had good public relation campaigns through print, radio, and television.

3. Integrations across regulatory bodies: Incentive-based policies should be communicated with all agencies or departments involved in neighborhood design and included in the regulations used by said bodies. For example, regulators, such as civil engineers that review stormwater plans, have a set of guidelines that they follow and are reticent to permit development plans that are unusual and are not addressed in the guidelines. LID treatment systems often include rain gardens or natural swales, and these features may not be permitted because the engineers are more comfortable with conventional "curb and gutter" approaches. Building government capacity is crucial in this case because if a green site plan is submitted to the regulatory division and they are not familiar with the LID practices, approval may actually take longer than a more conventional submittal. This may go against the original "incentive" which was to fast track the permitting process! Because there are many kinks and roadblocks in the permitting process that are unknown until somebody submits a plan, we suggest that county and city governments try out the policy on their own properties first before they make it available to the private sector. This way, a smoother process can be worked out before a private individual or entity submits a development plan.

When crafting policies for protecting wildlife habitat, more rigorous ecological guidelines are needed for land use regulation and design. Particularly, policies must address design, construction, and postconstruction issues. In a recent review of local land use regulations that establish guidelines or create incentives for conservation development, we found that the conservation design and consultation requirements are relatively weak (Reed et al. 2014). For example, although it is a basic principle of the ideal conservation design process, ecological site analysis was required by only 13 % of conservation development ordinances adopted by western US counties, and ecological site analysis was required to be completed prior to developed area design by only 5 % of ordinances. In addition, only 8 % of conservation development ordinances require consultation with an ecological expert or plan in the design of the protected open space, despite the fact that knowledge of and capacity for biodiversity conservation is highly limited among local planning jurisdictions in the USA (Miller et al. 2009). Beyond the land use regulations themselves, there is considerable room for variability in implementation, through variances in the review process, as well as in the construction and postconstruction phases of development (Vale and Vale 1976; Hostetler and Drake 2009). Second, policies should address the coordination of multiple conservation developments across a region. Broader-scale processes would benefit from coordination of project locations and protected area design and management through reference to a landscape-level conservation plan (Arendt 2004; Freeman and Bell 2011). In summary, there is a strong need for conservation expertise regarding how best to design and steward a development project, and that is a role that wildlife ecologists could help to play.

Having said the above, if scientists and city/county staff collaborate with a developer to build conservation developments, these local examples go a long way to help others see that there is a different way to build communities. It is much more digestible (to reluctant adopters) when people can see unique design and management strategies. Further, a local example helps create a planning and permitting process that is progressively easier for adopters over time. For example, a green development in Gainesville, Florida (called Madera—see http://www.wec.ufl.edu/extension/gc/madera/) helped to introduce local officials to LID practices such as narrow roads, swales, rain gardens, permeable pavement, and wooded stormwater retention ponds. The project experienced a number of difficulties getting these relatively new stormwater practices approved by local regulators. Although not all of the practices were given proper stormwater credits, this project started a conversation among county and city regulators and has helped LID practices to become more mainstream in Gainesville. Another example is a green development in Davidson, North Carolina (called Woodlands at Davidson) where the developer piloted several LID practices. This project started a conversation with local regulators and planners, which resulted in policy makers adopting a new LID ordinance (Allen et al. 2012). We cannot overstate the importance of local examples that showcase new technologies, management strategies, and designs. Such projects provide a platform for developers to see what could be done (and that it is not difficult) and it helps ferret out local regulations that initially may have been barriers, making it easier for the next conservation development project to be implemented (Hostetler 2012b; Allen et al. 2012).

13.4.2 Research Priorities for Conservation Development

Although the scientific literature on the effects of urbanization on wildlife continues to grow (Magle et al. 2012; Chap. 1), few empirical field studies have been conducted in the context of conservation developments. Conservation developments impact wildlife both within the boundaries of developments and across large regions. Priorities for future research on conservation developments should address both human and ecological dimensions. At a basic level, we need a better understanding of how housing density and configuration in conservation developments influence wildlife species. How densely should homes be clustered, and where should housing clusters be located on a development property? Likewise, we need a better understanding of the effects of conservation design choices. What proportion of the site should be protected as open space, which natural features are most important to protect, and how should the protected land be configured? Design of both the protected and developed portions of a conservation development are influenced by scale and landscape context. How do the size of the property and its development yield (i.e., number of lots or homes per acre) influence outcomes for wildlife? How are wildlife species within conservation developments affected by human land use and activities on adjacent properties? Do benefits to wildlife increase when conservation developments are located and designed according to a regional conservation plan or green infrastructure plan (Benedict and McMahon 2006)?

Moving beyond design factors, research priorities should also include construction and postconstruction issues, such as investigating how land management practices influence wildlife populations over time. Do conservation outcomes differ by land protection instrument (e.g., a conservation easement, restrictive covenant, or collective ownership)? Who owns and stewards the conserved areas of a development, and what is their expertise regarding wildlife management? What are the influences of permitted activities (e.g., agriculture, recreation) or informal use of the protected lands by residents? Studies of management and stewardship should further incorporate human land use practices on private lots. For example, what are the effects of landscaping, water and resource provisioning, management of refuse, domestic animals, light and noise pollution?

In addition to being a model system for investigating the effects of residential development, conservation developments are natural subjects for interdisciplinary research. Currently, studies integrating biophysical with social or economic factors comprise only a very small proportion (2.1 %) of research on the effects of residential development in the United States (Pejchar et al., unpublished data, Chapter Chap. 3). However, the economic viability and social sustainability of conservation developments are likely to have a substantial influence on how widespread the development practice becomes and may also contribute to their effectiveness for protecting wildlife. Research studies should be designed with reference to realistic regulatory, design, construction, and stewardship alternatives.

Finally, social research that addresses how to communicate with the public, builders, and policy makers is needed if best design and management practices are going to be adopted and actually used in conservation developments. Surveys of various stakeholders have demonstrated confusion and limited knowledge about the benefits and costs of conservation developments and low impact developments (Bowman et al. 2012; Allen et al. 2012). Key research questions to address are: (1) How can scientists effectively engage with public and private entities? (2) How can city and county planning be reformed to incorporate sustainable designs and management? (3) What communication strategies work for different stakeholders? Without a thorough understanding of mechanisms to engage stakeholders in implementing conservation developments, all the best designs and practices will continue to reside in a published format among journals, and actual on-the-ground implementation will not be realized.

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Chapter 14
Managing Urban Wildlife Habitat at the Local Scale

Christopher E. Moorman

Keywords Vegetation composition · Vegetation structure · Early succession vegetation · Vegetation management · Native plants · Invasive plant removal · Recreational trails · Domestic pets · Cats

Homeowners and managers of municipal properties can make a significant contribution to urban wildlife conservation in backyards, neighborhood common areas, or local parks (Goddard et al. 2009). Although only 2.8 % of the earth's surface, the urban land base comprises more than 41.5 billion ha, yielding ample space to create and maintain habitat for wildlife (Millennium Ecosystem Assessment 2005). A majority of the urban land base lies in suburban and commercial developments, where individual home or business owners have local control over many of the factors (e.g., establishment and maintenance of vegetation in the landscape) that influence wildlife habitat. Herein, I provide a road map to successful management of wildlife habitat at the local scale, beginning with several guiding principles that should hold true in nearly every urban area around the globe. I describe key challenges to managing local habitat patches in the urban landscape and strategies that should improve the likelihood the habitat is managed appropriately for target wildlife. Most of these approaches involve protecting or restoring vegetation communities. Where vegetation has been mostly or entirely replaced by the built environment and associated impervious surfaces, opportunities to manage habitat locally will be limited. On the other hand, where large expanses of natural area were protected, relatively low levels of management may be required to conserve wildlife. Because every habitat patch has a unique management history, vegetation composition and structure, and surrounding context, each deserves a unique plan for its conservation.

C. E. Moorman (✉)
Fisheries, Wildlife, and Conservation Biology Program, Department of Forestry and Environmental Resources, North Carolina State University, 110 Brooks Avenue, Box 7646, Raleigh, NC 27695, USA
e-mail: chris_moorman@ncsu.edu

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Finally, I address strategies to mitigate the potential detrimental effects of human structures that are common in urban areas (e.g., windows, communication towers, and power lines).

14.1 Guiding Principles

Habitat management in the urban environment will be most effective if conducted with consideration for four key factors that influence use of habitat patches by wildlife: (1) composition of the plant community; (2) species richness of the plant community, especially the prevalence of native plants; (3) vertical and horizontal structure of the vegetation; and (4) successional stage of the plant community. Each wildlife species has a unique set of requirements, and no single habitat patch can provide usable space for all local wildlife species. Therefore, management efforts should target the habitat requirements of focal wildlife species so resources are used efficiently and conservation goals are reached.

Composition of urban plant communities affects the distribution, amount, and timing of food and cover availability and, therefore, influences which wildlife species use habitat patches and when. For example, some animals associate with conifers (e.g., pine warbler (*Setophaga pinus*)), whereas others most commonly associate with hardwoods (e.g., eastern gray squirrel (*Sciurus carolinensis)*). Additionally, a diverse plant community ensures that food and cover are available year round and that alternative food sources, also known as buffer foods, are available when primary foods are not. Each plant species flowers and fruits at a different time, so greater diversity of plants generally results in a greater length of time that food resources are available to wildlife. Some plant species produce fruit crops in a cyclical pattern so that there can be boom and bust years, as is the case with acorn production by some species of oak (*Quercus* spp.). In the bust years, animals that depend on that food source must seek alternative buffer foods, which are likely to be more available in patches with a greater diversity of plant species.

The conservation of native plants is especially critical for wildlife in urban landscapes. For example, studies have shown that density and diversity of native birds in urban landscapes are greatest in areas with greater cover and volume of native plants (Mills et al. 1989; Germaine et al. 1998). Conversely, abundance of nonnative bird species may be greatest in areas dominated by nonnative vegetation (Germaine et al. 1998). Insects, especially lepidopteran caterpillars, are linked closely to native host plants, and many are absent when urban areas lack the native host plants (Burghardt et al. 2009). Additionally, avian nests constructed in nonnative plants in the urban landscape may experience greater rates of failure than nests placed in native plants (Schmidt and Whelan 1999; Borgmann and Rodewald 2004). These are just a few examples of problems posed for wildlife by increasing coverage of nonnative, invasive plants, and ongoing research is likely to expand our understanding of these relationships.

In forested systems, wildlife diversity tends to increase with the complexity of vertical vegetation structure. Vertical structure is the distribution of different layers of vegetation, including the ground layer, shrub layer, midstory layer, and canopy layer. Many wildlife species, especially birds, are able to distribute themselves vertically among these layers in a forest. Indeed, temperate hardwood forests with more forest layers tend to harbor a greater diversity of bird species (MacArthur and MacArthur 1961). However, well-developed canopy and midstory layers shade forest understories, thereby eliminating food and cover near the ground where most wildlife species persist. Canopy openings maintained via natural disturbance or management activities can help encourage development of the shrub and ground layers, thereby improving the habitat suitability for wildlife of lower forest layers (Moorman and Guynn 2001; Bakermans et al. 2012). In short, multi-dimensional plant communities provide more niches for wildlife than lawn and other two-dimensional plant communities.

Many specialist wildlife species are linked to specific stages of succession. For example, indigo bunting (*Passerina cyanea*) and least shrew (*Cryptotis parva*) commonly are associated with early succession plant communities dominated by herbaceous plants and shrubs. Protection, restoration, and management of early successional meadows and grasslands in urban areas help conserve unique plant and animal species. Conversely, salamander species typically are most abundant in late-succession plant communities, where the tree canopy shades the forest floor that contains a thick layer of leaf litter and cool, moist microclimate. However, these habitat types rarely are conserved in urban areas and both early and late seral stage specialists often are absent there; instead, the majority of wildlife species occurring in the urban landscape are generalists able to persist across a range of fragmented habitat types and successional stages.

The success of efforts to restore, create, or improve vegetation communities locally will be affected by distribution and management of habitat at the neighborhood (Chap. 13) and landscape scale (Chap. 12). For example, a small, isolated habitat patch surrounded by commercial development will not sustain area-sensitive wildlife species no matter how well the patch vegetation is managed. Conversely, large-scale efforts to conserve wildlife habitats in the urban landscape will not reach their full potential if conserved green spaces are not managed in a manner that maintains vegetation conditions appropriate for target wildlife species. Therefore, habitat conservation for urban wildlife must occur in an integrated fashion across every scale to be most successful (Goddard et al. 2009).

14.2 Managing Green Space

Protecting land from development provides a direct benefit to wildlife by conserving wildlife habitat, but management actions will be necessary to maximize the value of these natural areas. For example, hand removal or herbicide applications can be used to limit the spread of nonnative plants or canopy openings can be created or maintained to increase vertical and horizontal vegetation structure.

14.2.1 Using Disturbance as a Management Tool

Urbanites often attempt to limit anthropogenic and natural disturbance (e.g., fire suppression, pest insect management, tree care, few timber harvests), because they perceive it as detrimental to wildlife. Yet, all wildlife are well-adapted to disturbance and a majority of animals benefit from the habitat conditions created or maintained by some degree of disturbance (Hunter et al. 2001). In urban areas, species that use vegetation communities created or maintained by disturbance suffer because appropriate habitat management either violates local ordinances or norms for neat and orderly landscaping. For example, commercial timber harvest is unpopular in part because it often leaves woody debris considered unsightly. Increasingly restrictive air quality standards and the complexities of managing smoke make the use of prescribed fire a challenge, especially near areas of concentrated human dwellings. Many local governments also have land use regulations, including zoning or tree protection and landscaping ordinances, which can restrict the cutting of trees or disturbance of vegetative buffers. These regulations ultimately limit the use of timber harvesting and prescribed burning as conservation tools when managing for wildlife. Managers can work with local planners to ensure that zoning ordinances and other land use regulations do not restrict habitat management practices beneficial to wildlife in urban settings, but options for habitat management in the urban setting are likely to be less flexible than in more rural areas.

Where allowed, timber harvests and prescribed burning can be used to manage succession and maintain habitat conditions for disturbance-dependent plants and animals. Periodic (every 1 to 2 years) disturbance (e.g., mowing) of grassy areas or the margins of ball fields or other open areas will prevent woody encroachment and yield weedy or shrubby vegetation used by early succession wildlife. Similarly, restored prairie or a planted wildflower meadows that contain a diversity of native plants may be aesthetically pleasing to local residents, in addition to providing habitat for grassland wildlife (Lindemann-Matthies and Bose 2007).

The location and type of management for early succession vegetation, however, can be modified for urban contexts. Early succession vegetation communities can be maintained along utility rights-of-way or power line corridors, especially with the appropriate frequency and type of disturbance; often, this may happen unintentionally as a by-product of vegetation management to protect access. Conservation of larger, more contiguous habitat patches allows more cost-efficient implementation of management activities, such as prescribed burning and timber harvest. Similarly, special zoning areas with no development or low density development around green spaces can help buffer adjacent areas of higher density development from the aesthetic effects of timber harvest or fire and the smoke generated during prescribed burning. In forested natural areas, the use of intermediate timber management activities (e.g., thinning) or regeneration harvests (e.g., group selection harvests or shelterwood harvests) that leave standing canopy reduce aesthetic effects of timber harvest as compared to clearcuts. Finally, mowing and herbicides can be used to manage vegetation where prescribed fire and timber harvest are not an option, although these practices may not yield the habitat conditions required by some wildlife or plants.

14.2.2 Minimizing the Effects of Recreational Use

User-related impacts on open space resources include loss of soil and ground vegetation, soil erosion along trails, tree damage, and wildlife harassment (Cole and Landres 1996). Impacts on wildlife can be either indirect through habitat alteration or direct as excessive noise or disturbance from people and their pets. In Colorado, specialist birds (e.g., western meadowlark (*Sturnella neglecta*) in grasslands, western wood-pewee (*Contopus sordidulus*) in forest) were less common and nest predation rates were greater near recreational trails (Miller et al. 1998). Conversely, Miller and Hobbs (2000) documented lower rates of nest predation by mammals nearer to trails than in areas away from trails. Extensive clearing of vegetation along trails can improve sight lines for recreational uses but also breaks the forest canopy, essentially creating two narrower corridors that are less suitable to forest-interior birds; alternatively, recreational trails with narrow strips of managed vegetation did not affect forest-interior bird abundance (Mason et al. 2007). To avoid fragmenting forest habitat, trails can be consolidated along forest edges, as opposed to through the interior, or located in open areas (Miller et al. 1998; Mason et al. 2007). Additionally, visitor impacts on open space can be contained by closing sensitive areas to users or by concentrating use along designated trails (Leung and Marion 1999). Pets that accompany users should be leashed to prevent disturbance or predation of wildlife, and pet waste should be collected to prevent contamination of adjacent water bodies.

14.2.3 Managing Invasive Plants

Invasive, nonnative plants typically are more abundant in urban areas than in rural environments (Chap. 5). The colorful berries that make many nonnative plant species attractive as urban ornamentals also make them highly prone to become invasive. Birds and other wildlife commonly eat the fruits of nonnative plants such as olives (*Eleagnus* spp.) and privet (*Ligustrum* spp.) and disperse the plants great distances across the landscape as they defecate the seeds (Lafleur et al. 2007). Vegetation communities dominated by nonnative plant species typically are lower value to wildlife, especially urban avoider species, so prevention or control of nonnative plant invasions is important to conservation of urban wildlife habitat.

Accordingly, homeowners or other urban land managers can remove nonnative plants using a variety of methods. Where invasive plants are well established, complete elimination generally is not possible. In such cases, removal efforts must be balanced against the long-term cost, especially for species that spread quickly or where infestations are extensive. To save on costs and engage the local community in conservation efforts, volunteer groups (e.g., clubs, student organizations, and friends groups) can be enlisted to help remove invasive plants. Plants either can be removed by hand, mechanically (e.g., mowed, dug up, or pulled using a weed wrench), or with herbicides, and sometimes a combination of approaches may be necessary. However, removal efforts typically must be widespread and continuous to be successful, and therefore are costly over the long term (Kettenring and Adams 2011).

As with any risk, prevention is first principle of control. A majority of invasive plant species around the globe, especially woody plants, were first introduced by the horticultural industry (Reichard and White 2001). Therefore, an effective strategy to limit the spread of invasive plants is to avoid planting nonnative species known to be invasive or potentially invasive. Because there is a time lag between when a nonnative plant is first introduced and when it becomes invasive, identifying potentially invasive plant species can be challenging (Reichard and White 2001). Hence, an emphasis on native plants when landscaping for wildlife not only avoids introduction of known invasive plants but also helps limit the risk of nonnative species becoming invasive. Ultimately, efforts to reduce the introduction of invasive plants as landscaping ornamentals will require educating consumers, which in turn will help drive change in the nursery trade (Reichard and White 2001). These outreach messages should not only highlight the potential for nonnative plants to invade nearby natural areas, but also emphasize the benefits of native plants to wildlife.

14.2.4 Maintaining Dead Wood

Snags (i.e., standing dead trees) and downed coarse woody debris (CWD) are important features of forest ecosystems (Harmon et al. 1986). Diverse organisms, including invertebrates, vertebrates, fungi, and plants, require standing and downed CWD (e.g., Freedman et al. 1996; Hunter 1999). Downed CWD also may be important for nutrient retention (Harmon et al. 1986; Krankina et al. 1999) and water dynamics (Fraver et al. 2002). Because CWD could represent a long-term carbon reservoir in some forests, it has implications for atmospheric carbon balance and global climate change (Currie and Nadelhoffer 2002). However, snags decay and fall quickly in warm, humid regions, so snag loss must be balanced with constant recruitment (Moorman et al. 1999). In urban areas, snags are removed for safety reasons, because they are considered unsightly, or out of ignorance about the ecological values they provide. In turn, cavity-nesting bird densities are lower in urban areas than in natural areas (Tilghman 1987; Blewett and Marzluff 2005). To ensure the availability of snags to birds and other wildlife in urbanizing areas, Blewett and Marzluff (2005) recommended conserving patches of forest with the greatest densities of existing snags (>8 snags ≥25 cm DBH/ha). In addition to protecting snags that pose little safety risk, trimming dead trees to reduce the risk of deadfall onto valuable property and replacing lost snags with appropriate nest boxes increases habitat quality for cavity-nesting species (Blewett and Marzluff 2005). Efforts to conserve snags will need to be linked to educational efforts designed to convince planners, developers, and homeowners of the environmental value of dead wood (Blewett and Marzluff 2005).

Can Golf Courses Play a Role in Urban Wildlife Conservation?

Worldwide there are over 25,000 golf courses, with over 17,000 in the USA, more than 2600 in UK, and a rapidly increasing number in China, Japan, and Taiwan (Terman 1997; Cristol and Rodewald 2005; Tanner and Gange 2005). In the USA, golf courses average 55 ha in size and therefore provide substantial land area for conservation (Cristol and Rodewald 2005). In fact, numerous studies have demonstrated that golf courses harbor a high diversity and abundance of native wildlife, especially birds (Jones et al. 2005; White and Main 2005), and a study in New Mexico showed that native bird richness was greater on each of five urban golf courses than on the paired natural areas (Merola-Zwartjes and DeLong 2005). Intermediate disturbance can yield maximum levels of wildlife diversity, and support for this trend has been shown in studies of urban birds (Blair 1996). Similarly, golf courses represent an intermediate level of urbanization, greater than undeveloped natural areas and less than higher levels of urbanization dominated by human structures and impervious surface (Tanner and Gange 2005). Golf courses may include wetlands and structurally diverse planted vegetation, absent in surrounding urban and natural areas, especially when the native plant community is fairly simple (e.g., desert or grassland). Wildlife associated with wetlands may be especially abundant on golf courses that have substantial area of artificial ponds or riparian vegetation (Merola-Zwartjes and DeLong 2005; White and Main 2005).

Although wildlife present on golf courses typical are generalist species of relatively low conservation value (Cristol and Rodewald 2005), there are a variety of design strategies and management activities that can be used to increase the likelihood that urban avoiders are conserved on golf courses. Naturalistic golf courses that include substantial amounts of native vegetation communities are especially attractive to urban avoiders, including threatened species (Terman 1997; Merola-Zwartjes and DeLong 2005). There are large acreages of out-of-play areas on many golf courses, and conservation of critical habitat elements, including older, large-diameter trees with cavities, streamside vegetation, and native grassland, in these areas can provide habitat for native wildlife (Cristol and Rodewald 2005). Reduction of mowing, irrigation, and pesticide application in out-of-play turf can provide habitat for some grassland wildlife. Also, planting native trees and shrubs can increase plant species diversity and structural complexity of vegetation on golf courses, thereby improving the habitat quality for native wildlife. Clustering of remnant natural areas into larger habitat complexes and connecting these natural areas with habitat corridors can increase the size of habitat patches and facilitate dispersal by wildlife within golf course properties. However, the overall potential of individual golf courses largely depends on the context of the landscape surrounding the course, with greater potential for conserving native wildlife on courses surrounded by greater cover of undeveloped natural areas (Porter et al. 2005). Ultimately, acceptance of conservation practices by golfers and golf course managers is critical to success of conservation efforts, and use of interpretive signage can help facilitate education and buy-in by these stakeholders. Finally, Audubon International's (http://www.auduboninternational.org/acspgolf-program-overview) certification program for golf courses assists golf course managers with conservation of wildlife habitat and other environmental services (e.g., protection of water quality) and provides recognition for these efforts.

14.3 Managing Individual Home Lots

Individual home lots have tremendous potential for providing critical wildlife habitat if the preurban plant community can be preserved or if the property can be landscaped with a species rich and structurally diverse plant community. Further, managing vegetation on home lots to attract wildlife can increase property values while decreasing energy costs with careful selection and placement of trees (McPherson and Rowntree 1993; Donovan and Butry 2009). These energy savings, however, may be relatively modest in cities with temperate climates and high tree cover (Nelson et al. 2012). Additionally, backyard wildlife habitats can provide children a place to explore and develop a lifelong interest in wildlife and conservation (Louv 2005).

Attracting wildlife to home lots involves the same principles that guide management of larger green spaces. The process should start with by mapping and inventorying the property, making sure to identify where water, cover, and foods already are available. Special note should be taken of native trees or shrubs that provide fruits or seeds during at least some part of the year, nectar producing flowers, and old snags and stumps. Then, homeowners should decide what animals could prosper in their yards and determine what will need to be added or removed to support them. Homeowners should focus on native plants that provide food or shelter for target species (Table 14.1), and nonnative invasive species should be removed. Supplemental food, including bird feeders, and cover, including nest boxes and brush piles, can be added to complement the resources provided by the plant community.

14.3.1 Landscaping Principles

To conserve wildlife habitat, as much of the predevelopment plant community should be retained as possible during land clearing and construction of new home lots or commercial developments (Mills et al. 1989). During construction, contractors should protect residual trees and other native vegetation from fill dirt and heavy equipment. Contractors should use well-maintained silt fences to prevent silt from entering water bodies. Consideration of wildlife habitat should not end with the

Table 14.1 Recommended plants that are useful landscaping additions for wildlife in residential areas of North America. Check with local sources for native plants best suited to a local region. This table provides examples that might not be native or suited to all regions of North America

Kinds of plants	Wildlife needs fulfilled	Example species
Conifers	Conifers provide escape cover, winter shelter, and summer nesting sites. Some also provide sap, buds, and seeds.	Pines, spruces, firs, arborvitae, junipers, cedars, and yews
Produce fruits or berries from May through August	Provide foods for small mammals, thrashers, catbirds, robins, thrushes, waxwings, woodpeckers, orioles, cardinals, towhees, and grosbeaks.	Cherry (*Prunus* spp.), blackberry and raspberry (*Rubus* spp.), serviceberry (*Amelanchier* spp.), blueberry (*Vaccinium* spp.), grape (*Vitis* spp.), mulberry (*Morus* spp.), plum (*Prunus* spp.), and elderberry (*Sambucus* spp.)
Produce fruits or berries from August through December	Important foods for migratory bird fat reserves prior to migration and for nonmigratory species that need to enter the winter in good condition.	Dogwoods (*Cornus* spp.), blackgum (*Nyssa sylvatica*), buffaloberries (*Shephirdia* spp.), persimmon (*Diospyros virginiana*), spicebush (*Lindera benzoin*), and Virginia creeper (*Parthenocissus quinquefolia*)
Fruits that remain attached to the plants through winter; many are not palatable until they have frozen and thawed multiple times	Provide long-lasting foods for animals through the toughest periods of winter.	Chokecherry (*Aronia* spp.), crabapple (*Malus* spp.), hollies (*Ilex* spp.), snowberry (*Symphoricarpos* spp.), sumacs (*Rhus* spp.), viburnums (*Viburnum* spp.)
Nectar producing plants	Provide nectar that attracts hummingbirds, orioles, bees, and butterflies.	Flowers with tubular corollas attract hummingbirds. Other fruiting trees, shrubs, vines and flowers also provide nectar and sugars.
Grasses and legumes	Provide cover for small mammals and ground nesting birds—especially if the area is not mowed until after the peak of the bird nesting season. Some grasses and legumes also provide seeds. Legumes commonly are used as butterfly caterpillar host plants.	Native prairie grasses, including grammas (*Bouteloua* spp.), switchgrass (*Panicum virgatum*), and bluestems (*Schizachyrium scoparium* and *Andropogon* spp.), are becoming increasingly popular for landscaping purposes.
Mast or nut producing plants	Nuts and acorns are eaten by a variety of wildlife. These plants also provide tall plant structure and nesting cover.	Oaks (*Quercus* spp.), hickories (*Carya* spp.), buckeyes (*Aesculus* spp.), chestnuts (*Castanea* spp.), butternut (*Juglans cinerea*), walnuts (*Juglans* spp.), and hazels (*Corylus* spp.)

Table 14.1 (continued)

Kinds of plants	Wildlife needs fulfilled	Example species
Host plants for caterpillars, which are food for other wildlife	Caterpillars, other insects, and spiders are eaten by a variety of wildlife including birds, lizards, frogs, toads, and mice. These plants are known to attract the greatest number of caterpillar species.	Oaks (*Quercus* spp.), black cherry (*Prunus serotina*), willows (*Salix* spp.), birches (*Betula* spp.), blueberry (*Vaccinium* spp.), and elms (*Ulmus* spp.)

construction phase (Hostetler and Drake 2009). Where vegetation was cleared during construction, native plants should be planted to restore wildlife habitat. Any shared open space in a new neighborhood should be managed perpetually to sustain biodiversity, following the guidelines for green space management provided earlier in the chapter.

In addition to emphasizing native plants, landscaping activities should:

- Select the right plant for the right site by matching light and soil moisture condition requirements of plants with site characteristics.
- Include a diversity of native plants species known as hosts for butterfly and moth larvae, which are important food sources for birds (Tallamy 2007). Oaks (*Quercus* spp.), cherries (*Prunus* spp.), and willows (*Salix* spp.) are three types of trees known to serve as hosts for the greatest variety of caterpillars in the eastern USA (Tallamy 2007).
- Select plants that flower and bear fruit at different times of the year, thereby assuring fruits, seeds, and nectar will be available throughout most of the year.
- Cluster similar types of vegetation to allow wildlife easy access to seasonally abundant food sources without excessive movement and increased exposure to predators.
- Plant low-growing herbs and shrubs under taller shrubs and trees. This helps to provide the vertical complexity of the vegetation that is important to birds and other wildlife linked to specific vegetation layers. For example, different birds nest and feed in the ground, shrub, mid-story, and canopy layers of a landscape (MacArthur and MacArthur 1961). Other taxa, including reptiles, amphibians, and mammals, often are ground-dwelling and benefit from low shrubs and herbaceous plants.
- Avoid planting large-maturing trees and shrubs where they will overgrow their space and interfere with overhead utilities or crowd homes and other structures. Shrubs and trees should be at least 6 feet away from structures.
- Avoid planting fruit-bearing shrubs in medians or along roadsides; birds and other wildlife attracted to the food or cover provided by these plants may be killed by collisions with passing vehicles.
- Collect soil samples from different areas on a property, especially when soils have been altered significantly during the development process (local cooperative

extension centers often provide detailed information on how to proceed with a soil sample analysis).
- Minimize lawn coverage.

14.3.2 Managing Lawn

In 2005, more than 16 million ha of land were dedicated to turfgrass in the USA; this amount of land area was three times greater than dedicated to corn production (Milesi et al. 2005). Lawn cover occupies a substantial portion of land cover in urbanizing regions (>20 %), and lawn management contributes to the deposition of chemical pesticides and fertilizers at densities greater than agriculture (Robbins and Birkenholtz 2003). Movement of these pollutants into urban water bodies significantly degrades water quality and leads to eutrophication of aquatic systems. Further, irrigation of turfgrasses constitutes a significant portion of residential and commercial water use, especially in arid or semiarid regions where it can account for 75 % of household water consumption (Mayer et al. 1999). Because lawns commonly are managed as monocultures of single grass species, they are characterized by simplified vegetation structure devoid of food and cover; accordingly, lawns offer poor habitat quality for all but a select few ubiquitous wildlife species (e.g., American robin—*Turdus migratorius*).

Although complete elimination of lawn is unnecessary to conserve urban wildlife, reduction in the extent of turfgrass cover in exchange for increasing cover of native plants would increase the availability of food and cover for wildlife. Urban residents may be more receptive to this change than one might think, and in fact, Peterson et al. (2012b) showed that urban residents preferred a 50 % native plant garden over 100 % turfgrass. The study also demonstrated that many homeowners maintain extensive lawns because they inaccurately assume it's the social norm. Where lawn is maintained, fertilizer and pesticide application should be minimized. In addition, tolerating a diversity of plant species (e.g., clovers or native forbs) will increase the value of the lawn to insects and other wildlife while reducing maintenance costs.

14.3.3 Creating Brush Piles

Brush piles, although not appropriate for every urban lot, can be constructed to provide cover for songbirds (e.g., winter sparrows, towhees, wrens, and thrashers), rabbits, and other small mammals. Brush piles should be placed near food sources (e.g., bird feeders) or along travel corridors in the urban environment. They can be constructed from downed limbs, hedge clippings, or old Christmas trees. As the wood decays and settles, new material should be stacked on the pile. Before construction, homeowners should check that brush piles are not prohibited in their neighborhood landscaping ordinance. Homeowners also should be aware that a variety of animals, including those sometimes unwanted, like snakes, rabbits, and mice, may be as likely to use brush piles as songbirds.

14.3.4 Erecting Artificial Nest Boxes and Platforms

Nest boxes may be used as surrogates for natural cavities to provide nesting sites for a variety of wildlife species, especially birds such as great crested flycatcher (*Myiarchus crinitus*), screech owl (*Megascops asio*), chickadees (*Poecile spp*.), wrens (*Troglodytidae*), titmice (*Baeolophus* spp.), and bluebirds (*Sialia spp.*). Several considerations must be addressed when building and erecting nest boxes for wildlife.

- Nest boxes that benefit native birds should be designed with species specific dimensions for entrance holes, nest box interiors, and box depth.
- A well-designed nest box is made of sturdy lumber (e.g., pine, redcedar, or cypress wood), has a metal entrance guard to prevent expansion by woodpeckers (*Picidae*) or squirrels, and does not have a perch. Perches increase the use of nest boxes by aggressive birds like house sparrows (*Passer domesticus*) and European starlings (*Sturnus vulgaris*) and may limit use by native birds.
- Do not paint or stain the interior of the box. If you decide to paint or stain the exterior, use a nontoxic paint or stain. If painting, use a light colored paint (e.g., white) to allow the box to reflect, rather than absorb, radiant heat.
- To prevent easy access by nest predators (e.g., snakes and squirrels), next boxes may be placed on a wooden or metal post with a predator guard, or baffle, placed below.
- Boxes should be placed in a habitat and location appropriate for the target bird species.

14.3.5 Erecting Bat Houses

Bat houses are an excellent way to provide shelter for bats in urban environments, where natural tree cavities and other forms of cover can be limited. Proper roost temperature is probably the most important factor for a successful bat house. Interior temperatures should be warm and as stable as possible (ideally 80 °F to 100 °F in summer) for maternity roosts. Bat house construction should be tight, roosting partitions should be rough, and roosting crevices should be 1.9 to 2.5 cm wide (Tuttle et al. 2004). Plans for constructing bat houses are readily available on the internet or at a local library or wild bird store. Keys to occupancy involve temperature, location, and maintenance. Houses should come with instructions (appropriate to the region) on best exterior color of houses and how they should be located to receive adequate solar heating. Failure to consider the factors that affect the thermal environment of boxes accounts for more than 80 % of bat house rejection (White 2004).

14.3.6 Providing Supplemental Food

In the USA, over 50 million people feed birds, and bird-feeding is a critical means for people to remain connected to wildlife in urban settings (USFWS and US Census Bureau 2012). However, the effect of supplemental feeding on bird populations is variable and often debated (Temple 1988; Brittingham 1991). Potential positive effects include improved physiological condition resulting in greater overwinter survival, especially during extreme weather, and greater reproduction. Potential negative effects include greater rates of disease transmission among individuals at feeders, greater rates of accidental collisions with windows near feeders, or increased predation (Klem 1990; Brittingham 1991). Feeders also may favor granivorous bird species at the expense of insectivorous species, cause birds to shift their geographic ranges (e.g., allow birds to survive harsher winters and shift ranges north), or alter migratory patterns (Brittingham 1991). To minimize disease risk, feeders should be cleaned regularly, usually at least every 2 weeks, washing them thoroughly with hot, soapy water. Feeders should be incorporated into residential areas only as a complement to an appropriately designed landscape that includes a diversity of native plants and adequate cover. Placing feeders within close proximity to dense vegetation or brush piles provides feeding birds a quick access to escape cover.

14.3.7 Managing Free-Ranging Pets

Free-ranging domestic pets can have substantial effects on urban wildlife. For example, there are approximately 600 million nonnative and invasive domestic cats (*Felis catus*) introduced by humans around the world. Cats are well known for their ability to depredate native wildlife wherever they occur, especially in urban areas where their densities are greatest (Dauphine and Cooper 2009). Recent empirical evidence suggests free-ranging domestic cats kill 1.4 to 3.7 billion birds and 6.9 to 20.7 billion mammals in the USA annually (Loss et al. 2013). Domestic cats can cause local extinctions of wildlife in urban habitat fragments (Crooks and Soulé 1999). Additionally, cats and other domestic pets can serve as reservoirs and vectors for diseases and parasites that jeopardize populations of native wildlife (Dauphine and Cooper 2009). To reduce the detrimental impacts of cats and other domestic animals on native wildlife, pets should always be kept indoors, contained within a fenced environment, or placed on a leash. Indoor pets are less likely to be killed or injured by automobile collisions, depredated by coyotes (*Canis latrans*) or other predators, or injured by fights with other free-ranging pets. Feral cat colonies should be discouraged and free-ranging cats and dogs should be humanely captured and removed (Dauphine and Cooper 2009). Supplemental feeding by humans allows feral cat populations to remain at high densities, thereby increasing the negative effects on native wildlife and providing a source population of cats to disperse into nearby areas (Schmidt et al. 2007). A small but vocal group of cat colony caretakers and advocates dispute effects of outdoor cats on wildlife, and even believe feral cats are native wildlife (Peterson et al. 2012a). These groups have successfully institutionalized outdoor cat colonies in some areas despite opposition from the scientific community and local citizens (Lohr and Lepczyk 2013), so it is critical that the public knows about outdoor cats, their impacts, and any surreptitious efforts to legalize maintenance of colonies on public lands, or worse, on private lands that do not belong to caretakers.

14.4 Minimizing Strikes with Structures

14.4.1 Reducing Bird Collisions with Communication Towers

It is estimated that 7 million birds, primarily those that are migratory, are killed annually by collisions with communication towers and their associated infrastructure across North America (Longcore et al. 2012). Over a 29-year period, 44,007 individuals from 186 species of birds were collected under a single tower at Tall Timbers Research Station near Tallahassee, Florida, USA (Crawford and Engstrom 2001). And, the number of annual bird mortalities can be expected to rise as new and taller towers are erected, especially in urban areas. Therefore, it is important to consider several recommendations for reducing tower kills. Because mortality risk rises with increasing tower height, especially when guy wires are present, towers should be less than 100 m tall and should be constructed without guy wires (Longcore et al. 2008). Birds are most attracted to continuously illuminated lights, especially white lights (Longcore et al. 2008). Therefore, strobe or flashing lights are suggested over the use of steady-burning lights. Other suggestions to reduce bird collisions include co-location of new towers on existing towers or structures and avoidance of locating new towers in areas of extensive migrant bird activity (e.g., mountain ridge tops, boundaries of large water bodies).

14.4.2 Reducing Bird Collisions with Windows

Conservative estimates of annual bird mortality from collisions with windows in North America exceed 1 billion birds (Klem 1990; Dunn 1993). However, there are a number of strategies that can be employed to reduce bird collision with windows in the urban environment. Bird strikes with existing windows can be reduced by placing feeders close (within 1 m) to a window, and removing reflective vegetation from areas in front of windows (Klem et al. 2004, 2009). Additionally, planting trees and installing window awnings to block the sun from hitting the window may eliminate some reflection. Window screen, flash tape, and bird netting can be used to prevent birds from reaching the glass surface or to break up the reflection enough to direct flying birds away from the glass. Falcon or owl silhouettes attached to windows typically do not effectively reduce bird collisions, except when they reduce reflection of glass. In new construction, the proportion of glass should be minimized or less reflective glass material should be used. Also, angling windows downward 20 to 40 ° from vertical can aid in reducing collisions (Klem et al. 2004).

14.4.3 Reducing Wildlife Mortality from Power Lines and Power Poles

Birds commonly are killed from electrocution or collision with any of the over 65 million km of medium-high voltage power lines in use around the world (Bevanger 1994, 1998; Jenkins et al. 2010). In general, rapid flying birds with relatively large bodies and small wings (e.g., ducks and geese, rails, cranes, gallinaceous birds, and tinamous), or those often characterized as "poor fliers", are at the greatest risk of collision with power lines (Bevanger 1998; Janss 2000). In contrast, perching raptors are most prone to death from electrocution (Bevanger 1998). Electrocution takes place when a bird simultaneously touches two phase conductors and an earthed (i.e., grounded) wire; hence, larger species are more likely at risk of electrocution because the conductors and earthed wire often are far apart (Bevanger 1998). A comprehensive description of strategies to reduce avian mortality from interaction with power lines was published by Avian Power Line Interaction Committee (2012). Construction of new power infrastructure should undergo rigorous environmental review and the risks to wildlife should be considered. When possible, power lines should be buried (Jenkins et al. 2010). Additionally, new lines can be located in areas less likely to support collision-prone birds or birds of high conservation value; because many rare bird species are less likely to occur in urban areas, power lines in urbanizing regions may be less likely to pose a risk to these species. Removal of earthed-wire can reduce bird collisions and electrocutions, but this step is unlikely until economical alternatives for lightning conduction are developed (Jenkins et al. 2010). Marking lines to make them more visible or appear thicker seems to reduce collisions, but more extensive field testing of these approaches is needed (Bevanger 1994; Jenkins et al. 2010). Because birds often fly above tree-top height, power lines passing through forest can be situated below the height of the tree canopy to reduce risk of collision (Bevanger 1994). Wider spacing between lines can reduce the likelihood a bird contacts two lines simultaneously, and elevated perches or perching guards can remove perching raptors from electrocution risk by preventing contact with electrical wires (Bevanger 1994).

Conclusions

Efforts to protect and manage wildlife habitat locally in urban landscapes can yield substantial conservation benefits, especially when conducted in a coordinated fashion over larger spatial scales (Chaps. 12, 13). Efforts to maximize native plant

Table 14.2 Sample list of internet sites which provide unbiased information about attracting wildlife to residential areas. (Accessed August 2014)

Topic	URL
Landscaping for wildlife	http://web4.audubon.org/bird/at_home/rethink_lawn.html
	http://www.ncsu.edu/goingnative
Bird houses	http://www.ncsu.edu/goingnative
Bird feeders	http://www.ncsu.edu/goingnative
Birds and glass windows	http://digitalmedia.fws.gov/cdm/ref/collection/document/id/1431
All about birds	http://www.allaboutbirds.org/
Attracting bats	http://www.batcon.org/
Butterflies and moths	http://www.butterfliesandmoths.org/
Reptiles and amphibians	http://www.northeastparc.org/products/backyard.htm

species diversity and the structural complexity of vegetation communities tend to conserve a wider variety of wildlife species. Proactive efforts to design or retrofit wildlife-friendly urban structures, namely windows, telecommunication towers, and power infrastructure, can reduce wildlife mortality, especially for birds. However, these and other mitigation strategies can be costly and would benefit from technological advances that improve the ease and cost efficiency of implementation. Although many of these mitigation measures may be beyond the control of individual homeowners, there are many sources of information on the internet for people interested in ways to enhance their residential or commercial property for wildlife (Table 14.2). I caution against information from commercial vendors selling their products, who often may not have the best interests of wildlife or homeowners in mind. For example, farmhouse, windmill, and gingerbread bird houses are designed for consumer appeal and may be less attractive to target wildlife. There are so many information sources and great ideas that individual homeowners may be too overwhelmed to act. In the USA, the simplest starting point is to contact the National Wildlife Federation about the certified backyard habitat program, which is designed to help individuals or corporations plan and apply a wildlife habitat plan for a home site or small acreage (http://www.nwf.org/How-to-Help/Garden-for-Wildlife/Certify-Your-Wildlife-Garden.aspx). Similarly, the Wildlife Habitat Council in the USA provides a third-party certification program for corporate sites (http://www.wildlifehc.org/certification/). In summary, efforts to think and act locally contribute to global efforts to conserve wildlife in urbanizing landscapes.

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Chapter 15
Wildlife Friendly Roads: The Impacts of Roads on Wildlife in Urban Areas and Potential Remedies

Seth P. D. Riley, Justin L. Brown, Jeff A. Sikich, Catherine M. Schoonmaker and Erin E. Boydston

Keywords Edge effects · Roadside · Roadkill · Barrier · Underpass · Wildlife corridor · Wildlife fencing

15.1 Introduction

Roads are one of the most important factors affecting the ability of wildlife to live and move within an urban area. Roads physically replace wildlife habitat and often reduce habitat quality nearby, fragment the remaining habitat, and cause increased mortality through vehicle collisions. Much ecological research on roads has focused on whether animals are successfully crossing roads, or if the road is a barrier to wildlife movement, gene flow, or functional connectivity. Roads can alter survival and reproduction for wildlife, even among species such as birds that cross roads easily. Here we examine the suite of potential impacts of roads on wildlife, but we focus particularly on urban settings. We report on studies, both in the literature and from our own experience, that have addressed wildlife and roads in urban landscapes. Although road ecology is a growing field of study, relatively little of this research, and relatively few mitigation projects, have been done in urban landscapes. We also draw from the available science on road impacts in rural areas when urban case studies have not fully addressed key topics.

We considered urban roads to be roads within urban landscapes, defined as landscapes comprised of significant portions of commercial, industrial, and high-density residential development. However, many urban landscapes include areas with lower density residential development and natural open spaces. In fact, often urban

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S. P. D. Riley (✉) · J. L. Brown · J. A. Sikich · C. M. Schoonmaker
Santa Monica Mountains, National Recreation Area,
401 W. Hillcrest Dr., Thousand Oaks, CA 91360, USA
e-mail: seth_riley@nps.gov

E. E. Boydston
U.S. Geological Survey, Western Ecological Research Center,
401 W. Hillcrest Dr., Thousand Oaks, CA 91360, USA

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roads that pass through or next to open space are particularly important for wildlife conservation and management.

15.1.1 Types of Roads

Variation in the structure (e.g., size, construction materials) and traffic of roads can have profound impacts on how wildlife is affected by them. This variation can include presence of a median, composition of the median, presence and height of fences or walls, and in particular the nature of the surrounding land use (commercial, industrial, residential, agricultural, natural).

The largest roads with the fastest traffic are generally called freeways. Freeways are divided roadways with on- and off-ramps, no traffic signals, and high speed limits. Their size can vary from one or two lanes each way up to five or more. These roads eliminate significant habitat and represent a wide, bare, and formidable area for wildlife to traverse. Traffic volumes can be high on freeways, especially in urban areas. Freeways often require significant engineering and earth movement because of their size and the need for entrance and exit ramps, and they are often built far above or below the surrounding land. Elevated roads on posts facilitate wildlife passage underneath. However, if roads are built on soil or fill, even if vegetated, or sunken down below the surrounding area, they can represent an even greater barrier to wildlife than a paved road surface at grade. Highways are generally smaller than freeways, and typically have traffic regulation such as stoplights. However, highways can overlap significantly in attributes with smaller freeways or larger secondary roads, so the impacts of highways on wildlife largely reflect those of freeways or secondary roads depending on structural similarities. Highways and freeways are often divided, which can increase the overall width of the road corridor, increasing the barrier effects. Additionally, if the median is vegetated, it may draw animals to the road and increase the chance of mortality.

Smaller and generally less-trafficked roads that feed into highways or freeways are secondary roads. Because these secondary roads go through towns and cities, they often have significant development along them. The impacts of urban development immediately adjacent to secondary (or larger) roads interact with the impact of the road itself to further limit animal movement across the landscape. Where there is natural open space in cities, major secondary roads may go through or along natural areas. In these instances, roads have the potential to be significant sources of mortality (e.g., Riley et al. 2003). The incentive for animals to move across roads bisecting natural habitat is great, because animals may perceive habitat on the other side. Moreover, the lack of development along these road segments or the medians within them generally make them less of a barrier than larger urban freeways, but their traffic volume may still be high, contributing to the effects on survivorship.

In urban areas there are also tertiary roads that go through or between residential areas. These roads have relatively few driveways and may have speed limits >the 25 mph typical for residential neighborhoods. Because of their greater traffic volumes and speeds, these roads may often have a significant mortality effect, especially where they are bordering or traversing natural areas (Baker et al. 2007;

van Langevelde et al. 2009). These tertiary roads are typically just one lane in each direction, and while they may be barriers for smaller species, they are less likely to impede movement for larger carnivores and ungulates. Finally, there are also smaller residential roads, where the speed limit is generally 25 mph or less, driveways intersect the road often, and traffic is light. The least traveled of these are likely to be dead-end roads, or cul-de-sacs, where there is no through-access to other streets. In the USA, small urban roads that provide service between neighborhoods and arterial roads (tertiary roads) average about 4300 cars/day, and the urban arterial surface roads (secondary roads) average 15,612 cars/day (U.S. Dept. of Transportation 2011).

15.1.2 Road Effects

There are three broad ways by which roads impact wildlife. First, roads physically replace whatever was previously on the site. Typically roads replace some kind of open space and associated vegetation. This results in a linear area that may be quite wide, is no longer vegetated, and consists of unnatural substrate (asphalt, cement). So the road itself results in a loss of habitat and creates a large bare area, both of which can have a significant impact on the behavior of wildlife.

Second, the habitat and surrounding environment is significantly altered. In fact, Forman (2000) estimated that one fifth of the area of the USA was affected ecologically by roads. Vegetation is generally cleared immediately adjacent to the road, and fences or walls are often added. New, nonnative vegetation may be planted, or existing vegetation may be altered along the road. There may be a median between the directions of larger roads, which may be vegetated or fenced with guardrails or Jersey barriers. Road maintenance, chemicals applied to roadways (e.g., salt, herbicides, oil), artificial lighting, and traffic noise further alter the environment. For wildlife, many of these alterations are negative, and may extend far away from the road itself, although some effects can be positive for particular species.

Third, vehicles traveling on roads cause mortality for wildlife and affect wildlife behavior. Road impacts from habitat loss or alteration would still be present even if no cars were traveling on the road, but the vehicles themselves have a major impact. The extent to which vehicles affect wildlife communities depends on the speed, volume, and timing of the vehicle traffic.

All three of these things represent effects of roads in any landscape. However, here we are interested specifically in how urban roads affect wildlife, and how these impacts may be similar to or different from roads outside of cities in more rural or natural settings. For example, removal of habitat for an urban road may not be important if the road traverses an intensely developed area lacking wildlife. Conversely, if the urban road replaces even a small remaining patch of rare wildlife habitat, the impact may be disproportionately large. In terms of the surrounding habitat, the effects of the road may be particularly strong in urban areas if more light is needed for high-traffic volumes or tall sound walls are required to protect neighboring residential areas. On the other hand, the often vegetated and sometimes relatively natural areas along the road in an intensely urbanized landscape may provide valuable movement corridors that larger mammals can use to move through highly urban landscapes. Finally, traffic characteristics are different in urban areas. Traffic volumes are generally much greater than in rural areas, but speeds will often be lower. These higher volumes but lower speeds are likely to increase the barrier effects of roads, because animals are less likely to attempt to cross; however, wildlife mortality from vehicle collisions may be lower when speeds are reduced.

15.2 Impacts of Roads in Urban Areas

15.2.1 Road Mortality

Perhaps the most obvious and direct threat to wildlife from urban roads is the threat of mortality from being struck and killed by a vehicle. Within the USA alone, there are over 4 million miles of roads, with 27 % of these classified as urban roads (U.S. Dept. of Transportation 2011). In 2005, there were 1.9 million vehicle miles driven per mile of urban road as compared to 0.35 million miles driven per mile of rural road (Federal Highways Administration 2008), a more than 5-fold difference in the amount of traffic on average. The heavier traffic volumes and the high density of urban roads make them an important potential source of mortality for wildlife (Hodson 1965; Rosen and Lowe 1994; Fahrig et al. 1995; Lodé 2000). For many urban wildlife populations, wildlife–vehicle collisions (WVCs) are the primary source of mortality (Grinder and Krausman 2001a; Lopez et al. 2003; Cypher et al. 2009; Cypher 2010; Soulsbury et al. 2010).

Roads serve as a constant threat for many urban wildlife species which are forced to cross roads seasonally for dispersal or breeding, or more regularly (even daily) for animals that must move between habitat patches separated by roads (Atwood et al. 2004; Gosselink et al. 2007; Gehrt and Riley 2010). For many smaller species (such as reptiles, amphibians, and small mammals), it can be incredibly difficult to successfully cross roads, and mortality rates can reach 100 % (Aresco 2005). Larger and more mobile urban animals, such as carnivores or ungulates, are typically capable of crossing roads, but the high density and grid pattern of roads still create significant mortality risk by forcing large animals to make frequent crossings. Riley et al. (2006) documented that 52 % of coyotes (*Canis latrans*) and 40 % of bobcats (*Lynx rufus*) in an urban landscape near Los Angeles crossed major secondary roads. Interestingly, the risk of dying from a WVC was independent of how urban-associated (based on radio-tracking) animals were (Riley et al. 2003), and even animals living mostly within protected natural areas were at risk of mortality on roads passing through open space. Within urban landscapes, it may be more difficult for wildlife to access safe crossing points because roadside development has blocked at least one side of the road. In a study of 15 potential wildlife crossing points north of Los Angeles, Ng et al. (2004) found that the presence of natural habitat on both sides of the road was the most significant factor associated with use by bobcats and coyotes, although not for raccoons (*Procyon lotor*) which used all studied crossing points.

Some urban animals may be exposed to roads frequently enough that they learn how to cross them safely and may have reduced risk of WVCs, thus benefitting from the length of time they are able to survive along roads (Mumme et al. 2000; Erritzoe et al. 2003). Populations of species that flourish in urban areas may have developed ways to avoid WVCs through road avoidance, learned crossing techniques, or exploitation of roadside resources without attempting to cross. Sometimes animals that rarely cross roads are more vulnerable to mortality (Riley 2006), and coyotes in Chicago have been seen to look both ways before crossing streets (JLB, personal observation). More specific study of the reactions and vulnerability of individual animals to roads is needed (e.g., Grilo et al. 2012).

Many factors can affect the likelihood of WVCs in urban areas. Often wildlife is attracted to roads for resources on or near them, which can include garbage (Dill 1926), spilled grain (Forman and Alexander 1998; Boves and Belthoff 2012), insects (Jackson 2003), worms (Gouar et al. 2011), small mammals (Boves and Belthoff 2012), carrion (Finnis 1960; Hodson 1962; Fulton et al. 2008), and fruiting vegetation (Watts 2003; Dowler and Swanson 1982), some of which (e.g., trash) may be especially plentiful along urban roads. There are also other potential attractions along roads such as grit or salt (Mineau and Brownlee 2005; Leblond et al. 2006), basking areas (Lebboroni and Corti 2006), soft dirt for dust bathing, puddles for drinking and bathing, mud for nest building, and hard surfaces for breaking snails (Finnis 1960; Hodson 1962) that may result in WVCs. In a study of moose (*Alces alces*) in the city of Quebec, Canada, modeling indicated that removing salt pools from roadsides and placing them farther away could reduce moose–vehicle collisions (Grosman et al. 2011).

Road design can also impact the number of WVCs that occur. Roads that are raised or lowered or that have embankments may have altered rates of WVCs, although studies have shown some conflicting results. In some cases the WVCs rate was greater on roads that were level with the surrounding landscape (Pons 2000; Clevenger et al. 2003), while Lodé (2000) found that mammals were more commonly found on sunken sections of road while other vertebrates were found on raised roads (see also Grilo et al. 2012). Road width is important, particularly for slower-moving species, because wide roads take longer to cross. Urban roads are often wider to accommodate increased traffic. In general, wider roads also have greater traffic volumes, making it difficult to differentiate road width effects from traffic volume effects. Certain species, particularly smaller ones, avoid venturing across areas devoid of cover, making wider roads an important barrier for these species (Oxley et al. 1974; McGregor et al. 2008; Brehme et al. 2013), but potentially reducing mortality effects by preventing crossing attempts altogether.

Roadside and median vegetation can play an important role in road mortality for wildlife (e.g., Found and Boyce 2011). Tall roadside vegetation makes it difficult for motorists to see wildlife adjacent to the road (Grilo et al. 2009), but it can also be beneficial as it forces birds to fly higher and avoid vehicles (Clevenger et al. 2003;

Ramp et al. 2006), as well as causing drivers to drive more slowly (Lewis-Evans and Charlton 2006). Birds are most likely to be hit by vehicles when they forage, nest, or roost near roads (Hill and Hockin 1992; Erritzoe et al. 2003). Fruiting roadside vegetation has led to large numbers of WVCs in some cases (Watts 2003). During a 1-month period, 298 Cedar Waxings (*Bombycilla cedrorum*) were struck by vehicles while foraging on the fruit of silverberry *(Eleagnus pungens*) planted in the median of a four-lane highway in an urban area in Texas (Dowler and Swanson 1982). Orlowski (2008) found high vehicle mortality for birds where hedgerows and tree belts were adjacent to roads, and he suggested planting vegetation further away to reduce mortality.

Traffic volume is generally positively associated with mortality. Mortality for amphibians and reptiles is positively correlated with traffic volume and can lead to a reduction in adjacent populations (Fahrig et al. 1995). However, mortality can also decrease at very high traffic levels because animals are less willing to cross when traffic volumes increase above a certain level (Ng 2000; Clevenger et al. 2003; Riley et al. 2006). Many studies have found that mortality peaks at a certain traffic volume, and that at and above this level the road serves as a barrier (e.g., Seiler (2005) suggested 5000 vehicles/day was a barrier for Moose). In New Zealand, Brockie et al. (2009) reported that a traffic volume of 3000 cars/day served as a barrier and fewer animals were struck on the road. In a review, Charry and Jones (2009) found variability among studies, but that roads with traffic levels above 10,000 vehicles/day were complete barriers for all terrestrial vertebrate groups with a threshold as low as 6000 vehicles/day for certain groups, such as turtles. Reijnen et al. (1996) found that above 5000 vehicles/day, 7 out of 12 bird species had 12–56% lower population densities within 100 m of the road, but that above 50,000 vehicles/day all species were reduced. This study measured nearby densities instead of road mortality, but it may be that traffic volume as a barrier is less applicable for birds that are flying above and around the road, and greater volumes will continue to increase mortality.

In response to greater traffic volumes in urban areas, wildlife species may adapt their behavior to cross roads at night or during other times of reduced traffic volume (Kitchen et al. 2000; Grinder and Krausman 2001b; Riley et al. 2003; Bautista et al. 2004; Baker et al. 2007). Mortality rates may be greater where traffic volumes are more variable across seasons or throughout the 24-h day and night cycle, especially if the lower traffic volumes dip below the level necessary for the road to be perceived as a barrier.

For larger and faster species such as carnivores and ungulates, traffic speeds may impact WVCs rates more than traffic volume. In Sweden, 57% of all moose–vehicle collisions occurred in areas with a speed limit of 90 km/h, while areas with both lower and higher speed limits experienced fewer collisions (Seiler 2005). In the USA along Interstate 80 in Nebraska, average vehicle speed dropped from 106.2–122.5 km/h during 1969–1973 to 88.5–94.3 km/h during 1974–1975, with an associated 34% drop in road mortalities for all wildlife species combined (specifically medium to large mammals and pheasants (*Phasianus colchicus*; Case 1978)). Most species benefited from the reduced speed, but not all (e.g., white-tailed deer

(*Odocoileus virginianus*) and badgers (*Taxidea taxus*)). Even small changes in traffic speed can have dramatic impacts on wildlife. For example in Cradle Mountain-Lake St. Clair National Park in Tasmania, increases of only 20 km/h on the access road led to the local extinction of eastern quolls (*Dasyurus viverrinus*) and a 50 % reduction in Tasmanian devils (*Sarcophilus harrisii*; Jones 2000). After various tools were employed to return traffic to the original speed and deterrents were used to reduce the amount of wildlife crossing the road, eastern quolls were able to recolonize the area and their population recovered to 50 % of its former level. This sensitivity to traffic speed means wildlife stand to benefit greatly from the various "safe streets" initiatives aimed at reducing pedestrian and cyclist mortality by slowing and calming traffic (http://www.smartgrowthamerica.org/complete-streets). Even when speed limits are lowered in urban areas, drivers may travel at higher speeds at night, which could increase the threat to nocturnal wildlife (Ramp and Ben-Ami 2006), especially if animals shift their activity to night time hours in more developed areas (e.g., Riley et al. 2003).

Roads near wetlands and riparian areas can have dramatic impacts on semiaquatic vertebrate species and result in extensive road mortality and ultimately population declines (MacDonald et al. 1994; Ashley and Robinson 1996; Fahrig et al. 2001; Hels and Buchwald 2001; Aresco 2005; Shepard et al. 2008; Patrick and Gibbs 2010). Roads that bisect important migratory corridors without the presence of suitable fencing and culverts can be particularly detrimental to semiaquatic wildlife through WVCs sustained when attempting to access critical resources, such as amphibians traveling to breeding sites (Santos et al. 2007; Shepard et al. 2008). Certain populations of turtles appear to be attracted to urban wetlands, because these wetlands rarely dry up and are connected by culvert crossings allowing turtles to move safely between sites (Rees et al. 2009; Roe et al. 2011), indicating that semiaquatic populations can be protected even in urban landscapes.

Age, sex, and seasonality can all be important factors affecting the rate of road mortality. Boves and Belthoff (2012) found that barn owls (*Tyto alba*) were more susceptible to WVCs during the winter, likely because higher energetic demands cause the owls to spend more time hunting or feeding on or near roads. They also found that juveniles, which are more likely to disperse long distances, and females were more frequently killed by vehicles. Road mortality accounted for 40–50 % of all mortality in swamp wallabies (*Wallabia bicolor*) in a periurban area in Australia, and juveniles had a 50 % greater road mortality rate than adults (Ramp and Ben-Ami 2006). In a study of bats in Pennsylvania, Russell et al. (2009) found that nonreproductive females were more susceptible to WVCs than males, but that there was no apparent age bias. Seasonality of roadkill may vary by taxa; Clevenger et al. (2003) found that in Canada, mammals killed by vehicles were most frequently found in April, birds from May to August, and amphibians from June to August, with the majority occurring during two different rainfall events. In one of the most complete studies of road effects in an urban wildlife population, Lopez et al. (2003) found that male Florida Key deer (*O. virginianus clavium*) are more susceptible than females, especially in the fall, and deer that live closer to the road (US 1) are more vulnerable.

Mortality from vehicles is typically not thought to impact population density for common species (Hodson 1965), but it may have detrimental long-term effects on population health, and for rarer species it can add to natural causes of mortality and lead to a population decline (e.g., Florida Scrub-jays (*Aphelocoma coerlecens*), Mumme et al. 2000). Often, animals killed by vehicles are healthy (Sutton 1927) and in better condition than those that are killed by predators (Bujoczek et al. 2011). For endangered Key deer in Florida, vehicles were the main cause of mortality (>50%; Lopez et al. 2003), most vehicle mortalities were in good to excellent condition (Nettles et al. 2002), and population modeling indicated that South Big Pine Key, the more urban area with higher road mortality, was a population sink (Harveson et al. 2004). As road mortality is likely to be important for many urban populations and is likely to be an additive mortality source, it may often contribute to urban populations being less stable.

15.2.2 Road Effects on Movement and Gene Flow

Along with their effects on survival through WVCs, roads also affect wildlife by creating a barrier to movement. Barrier effects occur when animals are unable to approach or cross roads, turning continuous populations into subpopulations and potentially separating animals from valuable resources. Gene flow is inhibited when breeding individuals do not cross roads or are killed crossing roads. In urban areas, road networks become very dense, separating wildlife into small, disconnected populations that are vulnerable to extinction (Underhill and Angold 2000; Anderson et al. 2011). Many urban roads exhibit high traffic volumes that may exceed 3000–5000 vehicles/day, levels where roads often become barriers (Clevenger et al. 2003; Seiler 2005; Federal Highways Administration 2008; Brockie et al. 2009; Charry and Jones 2009). Accordingly, the barrier effect from roads may be one of the most important impacts on wildlife in urban areas.

For most species studied in urban landscapes, roads act as a barrier to movement and often as a home range boundary. The barrier effects can be especially strong for smaller species, but barrier effects are frequently present for large vertebrates as well. For example, bobcats and coyotes in urban southern California often maintain home ranges with roads (including freeways) or other development as boundaries (Riley et al. 2003, 2006). Hedgehogs (*Erinaceus europaeus*) in England never crossed large urban roads, and crossed small urban roads less than expected (Rondinini and Doncaster 2002). In some cases, even smaller rural highways may create significant barriers to movement and result in home ranges abutting the road. This was the case with some bobcats in northern California (Riley 2006) and with bobcats along a highway through an agricultural area in southern California (Brown and Riley 2014; Fig. 15.1).

Although freeways may form barriers for many animals including bobcats, coyotes, and mountain lions (*Puma concolor*; Riley et al. 2006, 2014), these species are capable of crossing medium-sized urban roads regularly (Riley et al. 2003;

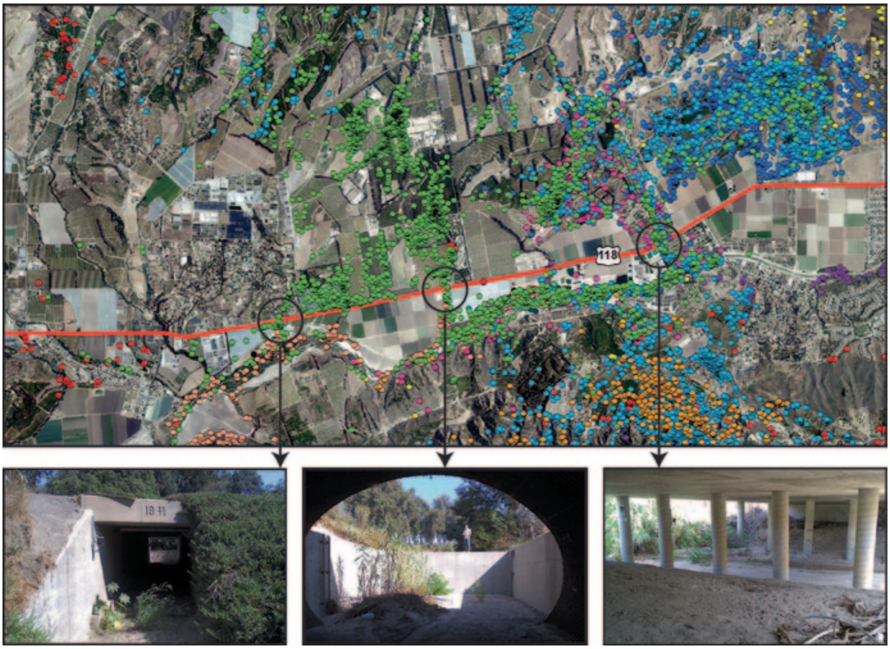

Fig. 15.1 Location data for 9 GPS-collared bobcats (individuals represented by *different colors*) that were tracked during 2012–2013 along State Route 118 in Ventura County, California, a two-lane highway. Even for this road that was relatively small and had relatively low-traffic volume, bobcats rarely crossed except by using three culverts: (from *left* to *right*) a box culvert, a round culvert in a dead-end channel, and a bridge underpass (Brown and Riley 2014)

Beier et al. 2010). Many species, including coyotes (Atwood et al. 2004; Gehrt and Riley 2010), raccoons (Hadidian et al. 2010), red foxes (*Vulpes vulpes*; Soulsbury et al. 2010), and bobcats (Riley et al. 2010), maintain smaller home ranges in urban areas. When those home ranges are largely within natural areas, smaller home ranges can mean that animals encounter fewer roads, despite the high road densities in cities. Additionally, some animals may shift their home range to avoid certain roads (Brody and Pelton 1989) or use their home range in ways that reduce the number of road crossings needed (Baker et al. 2007). These behavioral changes allow some mobile species to live and even thrive in urban landscapes despite high road densities and heavy traffic volumes (Grinder and Krausman 2001a; Prange et al. 2003; Gehrt 2006; Baker et al. 2007; Cypher 2010).

The barrier effect of roads can be especially severe for smaller, less mobile species. An early but striking example of the isolation effect that roads can have was demonstrated in a population of Malaysian house rats (*Rattus rattus diardi*) isolated in a small area (100 acres, 40 ha) completely surrounded by roads. Every rat captured in the population was infested with the mites that serve as the vector for scrub typhus. No mite infestation was detected in the adjacent populations in any direction (Lawley 1957). More recent and direct studies have shown that small mammals avoid crossing even smaller roads. Even dirt roads can be barriers for small vertebrates that avoid open areas, and small paved roads with little traffic can be strong barriers (Brehme et al. 2013). Many motorists in urban areas have experienced a squirrel (*Sciuridae*) suddenly darting in front of their car, but these crossings may actually be rare relative to overall movements by these animals. Studies show that rodents avoid roads, and that as roads increase in size, crossings are less frequent (Oxley et al. 1974; Clark et al. 2001; Rico et al. 2007; McGregor et al. 2008). McGregor et al. (2008) found translocated small mammals were capable of crossing roads, but they were less likely to return to their home range than animals not translocated across roads, and that the road itself served as the barrier to movement rather than traffic, emissions, or noise. More study is needed of small vertebrates and urban roads, but strong evidence that even small roads inhibit movement coupled with the high road densities in urban areas suggest that many of their populations face increased population isolation in urban landscapes.

Species that persist in urban landscapes may still be negatively affected from genetic differentiation and the loss of genetic diversity through drift and inbreeding associated with fragmentation from roads. In a review, Holderegger and Di Giulio (2010) found roads increase genetic differentiation and reduce genetic diversity, despite the fact that road barriers are relatively recent. In an early example, European common frogs (*Rana temporaria)* showed significant genetic differentiation and reduced diversity in populations separated by roads and railroads (Reh and Seitz 1990). Noël et al. (2006) found that eastern red-backed salamanders (*Plethodon cinereus*) exhibited significant genetic differentiation and reduced genetic diversity in habitat fragmented by roads in Montreal, while populations in contiguous habitat did not. Delaney et al. (2010) found genetic divergence associated with fragmentation, in a similar pattern, for three divergent lizard and one bird species, and the presence of major secondary roads between fragments was a significant factor associated with differentiation in all four. Even for larger, more mobile species, larger urban roads can act as barriers to gene flow. Riley et al. (2006) found genetic differentiation across a busy freeway (US 101) in southern California for both bobcats and coyotes after just 50 years or 25 generations, at least twice as much differentiation (as measured by F_{ST} values) across the freeway as between subpopulations on the same side. This same freeway is also a barrier to gene flow for mountain lions; in the small, isolated population in the Santa Monica Mountains south of US 101, genetic diversity is lower than anywhere else in the west, with the exception of an other isolated population in southern California (Riley et al. 2014). Recently, Serieys et al. (2014) examined the genetics of bobcat populations across a wider region surrounding the Santa Monica Mountains and confirmed genetic differentiation between bobcat populations across US 101, but also found a similar level of differentiation across another busy freeway, Interstate 405 (I-405). In this landscape in southern California, as in many urban areas, the freeways are corridors for development, so it can be difficult to distinguish between barrier effects related specifically to the road, to urban development, or to both (see also Sect. 15.3).

Mills and Allendorf (1996) recommended that a minimum of one to ten immigrants per generation need to reach an isolated population to maintain genetic diversity. For species that typically migrate or disperse each year (e.g., amphibians moving to breed), the migration of ten individuals is likely not enough to maintain sufficient reproduction or survival for overall population stability (Mansergh and Scotts 1989; van der Ree et al. 2009). For species such as large carnivores that typically occur at low population densities, it may not be feasible even in ideal situations to reach ten immigrants per generation, although in some cases one effective migrant can make a big difference (Vila et al. 2003; Riley et al. 2014). A further complicating factor is that the number of animals crossing roads is greater than the number of genetic migrants, as many dispersers may not successfully establish territories and reproduce.

Behavioral and social factors may play role in reduced connectivity across roads in urban areas by limiting both the number of migrants and the number that successfully reproduce. The most common type of behavioral barrier is the lack of willingness to cross unvegetated areas (McGregor et al. 2008) or other physical obstacles (e.g., walls or fences). However, because many species set up home range boundaries along roadways, for territorial species there may also be a social barrier to movement that forms in association with roads (Riley et al. 2006). Specifically, when animals encounter the territories of established animals, negative intraspecific interactions may cause the migrant to turn around or keep moving without successfully finding a suitable territory or mate. High densities of individuals located near roads likely strengthen the effect of the social barrier (Riley et al. 2006; Bissonette and Rosa 2009). The ability of animals to successfully cross roads, establish territories, and assist with the maintenance of gene flow is a function of the physical permeability of the road and the surrounding landscape and any behavioral and social barriers. Increasing the permeability of roads may reduce the influence of a social barrier, as increased ease of crossing and more crossing locations should help animals seeking to avoid conspecfics find a vacant territory to occupy.

15.2.3 Roadsides and Edge Effects of Roads (Negative and Positive)

On many highways, the area along the road, or roadside, is wide enough to be considered a distinct habitat type for some animals (Adams and Geis 1983; Taylor and Goldingay 2004; Orlowski 2008). The roadside, also known as roadside right-of-way (ROW), roadside edge, or roadside verge, is designed to uphold the physical integrity of the road and to provide motorists a safe place to pull over. ROWs may extend far away from the road and can vary in their value for wildlife. For example in agricultural areas, roadsides typically consist of exposed dirt edges with open ditches for rain runoff, while roads in forested habitat are surrounded by dense vegetation (Seiler 2001). In urban areas, plantings on roadsides are often designed to be both functional and aesthetically appealing. Planting vegetation along urban roads creates barriers that reduce traffic noise, create wind breaks, and reduce light disturbance for people who live near roads (Taylor and Goldingay 2004; Seiler and

Folkeson 2006). Roadsides in urban areas are generally not created for wildlife use, but the addition of these often well-vegetated corridors can contribute to the value of developed areas for wildlife by providing cover, food, and corridors for movement (Getz et al. 1978; Curtis and Jensen 2004; Fulton et al. 2008; Morgan et al. 2010). Although roadsides have some characteristics that are favorable for wildlife, associated habitat loss, noise pollution, light pollution, traffic, and pesticides can create a hostile environment that deters many species from approaching roads (Hill and Hockin 1992; Forman and Deblinger 2000; Kociolek et al. 2011).

15.2.3.1 Negative Edge Effects of Roads

15.2.3.1.1 Habitat Loss, Alteration by Edge Effects

The loss of natural habitat from construction and roadside maintenance can reduce wildlife activity in roadsides. Roadside vegetation often is trampled or removed during construction, but even after recovery, vegetation clearing adjacent to the road can greatly decrease the abundance and diversity of birds using roadsides, except for scavengers like common ravens (*Corvus corax*; Fulton et al. 2008). Along with reducing habitat, roads introduce artificial edges with sharp changes in vegetation types. Herbicide spraying and mowing along roadsides are used to prevent vegetation from blocking traffic signs, damaging road surfaces, and obstructing driver's views; ultimately they reduce tree cover and favor grasses. The openness of roads and the reduction in vegetation cover also create changes in the microclimate, often increasing temperatures along the road edge relative to more distant areas (Blouin-Demers and Weatherhead 2001). Mader (1984) observed microclimate changes (temperature, humidity) up to 30 m from the edge of a forest road. This change in temperature along the road edge may benefit some reptile species that need warmer areas for thermoregulation (see 15.2.3.2.1), but extreme changes in microclimate along roadside habitats may deter other species such as amphibians that are not able to tolerate desiccating conditions there (Langen et al. 2015).

15.2.3.1.2 Traffic Volume

High-traffic volume can create disturbance along roads and render roadsides unusable for some bird species (Reijnen et al. 1995). A study in Holland determined that disturbance from traffic had effects on three wading bird species: black-tailed godwit (*Limosa limosa*), lapwing (*Vanellus vanellus*), and redshank (*Tringa tetanus*). Traffic disturbance affected individual birds nesting at distances from 500 to 600 m from the road for a quiet rural road and up to 1600–1800 m away for a busy highway, amounting to a 60% reduction in population density that was related to the type of road (Hill and Hockin 1992). Large mammals can also be affected by traffic volume. Singer (1978) found that mountain goats visiting mineral licks showed reactions to vehicles passing on the highway. Grizzly bears (*Ursus arctos*) in Montana showed no or even positive selection for areas surrounding closed roads or roads used by <10 vehicles/day but avoided areas surrounding roads with >10 vehicles/day (Mace et al. 1996). In Arizona, elk (*Cervus canadensis*) were more likely to use habitat near the highway when traffic volumes were low (<100 vehicles/h). Elk shifted away from the highway during times of high vehicle traffic and returned to areas near the road when traffic volumes were lower (Gagnon et al. 2006).

In an urban landscape in southern California, Riley et al. (2003) found that bobcats and coyotes were often located in natural habitat during the day, and that both species were more likely to cross urban roads and venture into developed areas at night when traffic volume and human activity were lower. However, both bobcats and coyotes in this area still used natural habitat near roads even during the day, something that is difficult for them to avoid given the highly fragmented nature of the landscape. In an interesting case, Pescador and Peris (2007) found that predation rates were higher on birds nesting adjacent to roads with medium- to low-traffic volume than on birds nesting near roads with more traffic. They suggest that while predators generally avoid roads with high traffic, WVCs on roads with moderate or low traffic may attract predators and thus increase nest predation along these roads. Overall, however, the high traffic volumes in urban areas will decrease the value of roadsides as wildlife habitat.

15.2.3.1.3 Noise Pollution

One of the negative effects of increased traffic is noise. Traffic noise can cause frequent or chronic disturbance for some species. Forman and Deblinger (2000) report that traffic noise is the primary cause of avian disturbance in roadside habitat, and traffic noise can affect bird communities hundreds of meters from the road, with a reduction in bird abundance and diversity (Reijnen et al. 1995; Forman and Deblinger 2000). The reduction in bird density and diversity along noisy roadways may be in part explained by birds having difficulty communicating through calls and songs. Song sparrows (*Melospiza melodia*) will adjust their vocalizations to reduce masking from urban noise (Wood and Yezerinac 2006), but the small body size of some species may not allow them to produce vocalizations loud enough for conspecifics to hear over the din of urban environments (Brumm 2004). Some bird species are considered to be urban-exploiters; they are able to adapt to the urban environment and seem to be less affected by noise pollution and other urban disturbances. A study in Spain documented a positive relationship between urban-exploiter birds and roads. Urban-exploiter species contributed significantly to the high density of birds in cities and adjacent areas, but bird diversity was lower in these disturbed areas compared to natural areas far from development (Palomino and Carrascal 2006, 2007). Traffic noise can also affect communication of mammals by masking signal reception and creating false alarms. For example, the low frequency of vehicle noise closely overlaps with the frequency created by Stephen's kangaroo rat (*Dipodomys stephensi*) during footdrumming, so that deceptive signals or "false alarms" are generated by vehicle noise, and true alarms are masked (Shier et. al. 2012).

15.2.3.1.4 Light Pollution

Artificial light along roads from light poles, buildings, and car headlights can alter the night time environment for animals in roadsides. In general, the effects of light pollution on wildlife are overwhelmingly negative (Longcore and Rich 2004). Artificial light may increase nighttime foraging opportunities by attracting or making prey easier to see, but predators attracted to these altered conditions are at risk for WVCs. For example, nightjars (*Camprimulgidae spp.*) that have learned to associate roadside lights with flying insects are in danger of being hit by vehicles when they apply their sit-and-wait hunting technique to roadsides, and road mortality can be quite high for adult nightjars (Jackson 2003). Slow-moving toads may also become victims of road mortality as they forage on or near roadways for insects attracted to the light (Langen et al. 2015). Headlights can reduce the ability of animals on the roadway to escape vehicle strikes, as the lights can cause them to freeze in front of an approaching vehicle (Mazerolle et al. 2005). Light pollution is likely to be particularly relevant around roads in urban landscapes, because road lighting will be more extensive for safety reasons, and because commercial and residential development are often present around urban roads.

15.2.3.1.5 Chemicals

Road maintenance activities and vehicles can spread chemical pollutants into roadsides including salt, heavy metals, fertilizers, nutrients, and other toxicants (Seiler and Folkeson 2006). Roadside contaminants can be directly toxic for wildlife or can indirectly affect wildlife by degrading food and cover resources (Mineau and Brownlee 2005; Karraker et al. 2008). Heavy metals can accumulate in the tissue of plants and animals in roadsides, affecting reproduction and survival for small mammals (Scanlon 1987). Road salt used for deicing can have effects on aquatic systems as far as 1500 m away from the road (Forman and Deblinger 2000). Egg and larval survival decreased for spotted salamanders (*Ambystoma maculatum*) and wood frogs (*Rana sylvatica*) as salt concentrations increased in roadside pools (Karraker et al. 2008). The consumption of road salts can negatively affect some species, and salt toxicity may contribute to widespread mortality of birds around roadways (e.g., *cardueline finches*; Mineau and Brownlee 2005). Again in cities, specifically those at higher latitudes where deicing is common, the sheer density of roads is likely to make the problems of salt presence in roadsides severe.

15.2.3.2 Attractions of Roadsides for Wildlife

Roadsides are highly disturbed environments, yet some wildlife species are attracted to the resources available there. Planted areas adjacent to roads can provide nesting sites, shelter, and food (Haner et al. 1996; Palomino and Carrascal 2007; Fulton et al. 2008). In heavily altered landscapes such as urban and agricultural areas, roadsides may represent the only relatively natural habitat in the area and may also provide wildlife with suitable corridors to facilitate movement between fragmented natural areas (Seiler 2001). In Great Britain, roadsides support a great deal of the island's faunal diversity including 40 % of its mammal species, 100 % of the reptiles, 83 % of the amphibians, 20 % of the birds, and 42 % of the butterflies (Way 1977; Forman et al. 2003). Some nations consider roadsides to be of great conservation value and make efforts to protect and enhance natural habitat along roads; in Great Britain, six roadsides are officially designated as sites of special scientific interest (Forman et al. 2003).

15.2.3.2.1 Viable Habitat

Although habitat quality and structure vary throughout the world in roadsides, they can attract and support a variety of species (Baker 1971; Adams and Geis 1983; Bennett 1991; Curtis and Jensen 2004; Taylor and Goldingay 2004; Kociolek et al. 2011). Landscaped vegetation or disturbed grassy verges can increase species richness along roadsides by providing habitat different from the surrounding landscape (Adams and Geis 1983; Haner et al. 1996; Richardson et al. 2006; Palomino and Carrascal 2007). More specifically, shrubs and trees planted along roads can provide valuable nesting sites for birds and small mammals (Adams and Geis 1983; Orlowski 2008). Adams and Geis (1983) actually detected more small mammal species in roadsides than in the adjacent habitat, and determined that roadside habitat was attractive to grassland species but also to less habitat-specific rodents. The openness of roads creates changes in microclimate in roadsides that can provide ectothermic animals with increased opportunities for thermoregulation. For example, Richardson et al. (2006) found that female prairie kingsnakes (*Lampropeltis spp.*) used artificial edges along roads more often than natural edges. Raised embankments along roads may provide habitat in some urban areas, such as in Bakersfield, California where endangered San Joaquin kit foxes (*Vulpes macrotis mutica*) have been found to den in road embankments (Bjurlin and Cypher 2003). Numerous species, including bobcats, coyotes, raccoons, deer, gray foxes (*Urocyon cinereoargenteus*), skunks (*Mephitidae*), squirrels and other rodents, rabbits (*Leporidae*), various bird species, and even mountain lions (Schoonmaker and Riley 2011; Riley et al. unpublished data), utilize natural habitat adjacent to I-405 near Los Angeles (Fig. 15.2). In intensely urban landscapes such as around Los Angeles, wildlife may become habituated to freeway noise and activity, and in turn areas around freeways may represent important areas of habitat in a highly developed landscape.

15.2.3.2.2 Food and Nutrients

Roads may provide an increase in food resources for some wildlife species. Birds and small mammals are drawn to roadsides to forage on seeds that are more easily detected in short grass along mowed areas, berries and fruits from vegetation

Fig. 15.2 Locations of motion-activated cameras (*red stars*) that detected several mammal species, including a mountain lion (*left* photograph) and coyote (*right* photograph) in disturbed habitat along Interstate 405 (I-405) in Los Angeles County, California during 2012 (Riley unpubl. data). Just south of these cameras was one of two bridges over the interstate where camera monitoring showed animals occasionally crossing. (Schoonmaker and Riley 2011; Riley et al. unpublished data)

planted along roads, and invertebrates that are attracted to roads; birds will also collect sand and gravel found on roadsides as grit to help digest food (Laursen 1981; Dowler and Swanson 1982; Stapp and Lindquist 2007; Fulton et al. 2008). Morgan et al. (2010) found that roadsides can have a positive effect on food availability and energy intake for threatened Florida scrub-jays; scrub-jays handled more food items and spent less time foraging along roadsides compared to interior areas. The abundance of small mammals found in roadsides attracts raptors, and the presence of utility poles and fence posts along roads provides them with hunting perches (Hill and Hockin 1992). In open vegetated roadside areas, nutritious grasses, planted crops, and artificial salt pools are all attractants for ungulate species such as deer, moose, and mountain goats (*Oreamnos americanus*; Pletscher 1987; Laurian et al. 2008; Found and Boyce 2011; Clevenger and Huijser 2011). Salt is an important mineral for herbivores, and in areas where natural salt licks are not available, salt pools created by the deicing of roads are major attractants for ungulates (Fraser and Thomas 1982; Pletscher 1987; Laurian et al. 2008). Laurian et al. (2008) found that although moose usually avoided areas within 500 m of roads, some individuals would visit roadsides and feed on vegetation that contained higher levels of sodium than that in adjacent forest areas. In urban areas, roadsides planted with berry producing fruits can be an attractant for omnivorous mammals as well. In a study of coyote diet in an urban landscape in southern California, we found that coyotes consumed a high percentage of fruits from ornamental plants that could be found throughout the urban landscape including in yards, urban parks, and along roadsides (Gehrt and Riley 2010).

15.2.3.2.3 Corridors

In some instances roadsides may function as corridors (e.g., James and Stuart-Smith 2000), and in relatively hostile landscapes that are difficult to navigate, including densely urbanized ones, these roadside corridors could facilitate dispersal between fragmented natural areas. Meadow voles (*Microtus pennsylvanicus*) dispersed over 100 km in 6 years along grassy roadsides in Illinois (Getz et al. 1978). In the Netherlands, bank voles (*Clethrinomys glareolus*) traveled along wooded verges of rail and motorways to colonize the Zuid-Beveland peninsula (Seiler 2001). Young hedgehogs in England regularly dispersed along roadsides (Rondinini and Doncaster 2002). Road surfaces may even facilitate dispersal and have led to range expansion for some species. Cane toads (*Rhinella marina*), an introduced species to Australia, moved into previously inaccessible areas via roads (Seabrook and Dettmann 1996). Although roadsides may facilitate dispersal and movement for some species, they may also be challenging to navigate for long distances because road maintenance, construction, and embankments can form barriers along the way. Perhaps most importantly, road corridors are likely to intersect other roads, especially in urban areas, forcing animals to risk crossing the perpendicular road or creating a barrier for species not willing to cross. The use of roadsides as corridors may be more common for species that are less affected by the disturbance and habitat alteration present there.

15.3 Planning and Placement of Roads in Urban Areas

To accommodate urban growth, cities modify and expand their road networks, increasing the challenges for wildlife persistence in urban areas. However, including considerations for wildlife as early as possible in urban transportation planning can help mitigate impacts to animal populations. The most important opportunity to influence the outcome for wildlife lies in determining the path of the road across the landscape. Minimizing further habitat fragmentation from new roads may be particularly important for wildlife in urban areas where open space is already at a premium. One strategy to minimize fragmentation is “bundling” of roads—placing new roads near existing roads or other development rather than constructing new road paths through open space (Jaeger et al. 2005; Levin et al. 2007). This strategy may result in roads that trace the edges of the urban-wildland interface or in a wider road footprint if a new road runs beside an existing one. Another approach is limiting the length of road that bisects natural habitat by moving it to the narrowest part of the natural area. However, a tradeoff is that all of these approaches could lead to longer, more sinuous roads and an increase in road density (Gibbs and Shriver 2002; Rytwinski and Fahrig 2007).

Modifications of existing roads may offer ways to meet transportation needs without additional habitat fragmentation, but increasing traffic carrying capacity of roads may increase their barrier effects. Perhaps the most common form of mitigation for the barrier effects of roads is to provide safe passage for animals under or over roads while preventing them from accessing the road's surface (Clevenger and Waltho 2005; Grilo et al. 2008; Glista et al. 2009). Sites targeted for mitigation often include areas where habitat abuts both sides of the road, areas where high numbers of animals are hit and killed by vehicles (Barnum 2003; Grilo et al. 2008), or where roads isolate subpopulations of a species (Riley et al. 2006; Delaney et al. 2010). Predictive models for the impacts to wildlife can also aid in the selection of locations for crossing structures in the design of new roads. Because mitigation costs can be very high, particularly for new construction of underpasses and overpasses, prioritization and improvements for wildlife may often be a compromise between the ideal solutions for wildlife and available resources. However, there should be reasonable confidence that any measures undertaken will have value for wildlife and meet goals.

Changes to existing roads may present opportunities to insert mitigation measures for wildlife that were absent from the original design of a road or to offset negative impacts to wildlife when the traffic carrying capacity of a road is increased. For example, installation of wildlife fencing and new wildlife crossing structures were included as part of a substantial widening of State Route 71 in the Los Angeles-Orange County metro area. The original, narrower road had several culverts for drainage purposes, and the widened road included structures designed for wildlife use. A study before and after these changes found that connectivity for bobcats and coyotes did not decrease after freeway expansion and may have improved despite the widening of the highway (Alonso et al. 2014). Similarly, widening of the I-405 Freeway in Los Angeles included wildlife friendly modifications of a road bridge to decrease barrier effects and facilitate wildlife movement (Schoonmaker and Riley 2011).

Integrating road planning across the larger urban landscape can help a community meet transportation infrastructure needs while limiting adverse impacts to ecological connections for wildlife. Landscape connectivity plans or linkage maps that show priority areas for connectivity, such as Habitat Conservation Plans, exist for some large regions facing rapid development (e.g., Beier et al. 2006), and methods for designing linkages can be generalized to other areas (Beier et al. 2008). Furthermore, such planning that is often done for vast landscapes may be "downscaled" from broad regional plans to smaller urban areas that are more constrained by development (e.g., Thorne et al. 2006; Balkenhol and Waits 2009). Competing and conflicting interests between transportation and wildlife conservation goals, and potential cost limitations, are challenges to wildlife friendly improvements, but reducing WVCs and maintaining natural, intact areas of open space in cities are examples of outcomes that can benefit both humans and wildlife.

15.4 Mitigation of Negative Road Impacts in Urban Areas

As the science of road ecology has evolved, so have the collective efforts by public agencies and environmental stakeholders to mitigate the negative effects of roads on wildlife (Beckmann et al. 2010). These mitigation measures attempt to minimize the harmful ecological impacts of roads on wildlife populations, although mitigation alone cannot eliminate the detrimental impacts of roads. The most successful conservation efforts reduce WVCs and increase connectivity for a range of species. Despite the prevalence and importance of roads in urban landscapes, there are actually few studies explicitly studying mitigation efforts for roads in urban areas. Where possible, we have drawn on our own experience with wildlife and roads in southern California.

One of the most critical aspects of any effort to reduce the impacts of roads is selecting their location. There are various methods for producing connectivity or linkage maps (Beier et al. 2008). A critical element for any mitigation measure is that natural habitat remains on both sides of the road, and in many urban areas it is rare to have natural area abutting freeways or other major roads on one side, let alone both. For example, for the US 101 Freeway in southern California which separates the Santa Monica Mountains from open space to the north, there are very few places left where it is feasible to place crossing structures (Fig. 15.3). Considering wildlife connectivity needs and mortality risks before urban landscapes reach this point allows more options, at considerably less political and monetary expense.

15.4.1 Facilitating Movement: Crossing Structures

Wildlife crossing structures allow animals to safely move across a roadway, therefore maintaining habitat and population connectivity, reducing WVCs, and increasing motorist safety. Wildlife crossings come in a variety of shapes and sizes and include tunnels, culverts, and bridge underpasses going under roads, and road, wildlife, and canopy bridges going over roads (Forman et al. 2003; Glista et al. 2009). Crossing structures have been engineered and incorporated into road construction and improvement projects for road mitigation in many parts of the world (Forman et al. 2003). Many, if not most, crossing structures were not originally designed to facilitate wildlife movement (e.g., drainage culverts), but they have been found to benefit a variety of species (Haas 2000; Clevenger et al. 2001a; Ng et al. 2004).

Different types of animals require different wildlife crossing structures. For example, small mammals may prefer to use pipes or small culverts, while some ungulates and large carnivores select vegetated overpasses. In one study, grizzly bears, deer, and elk used overpasses more than underpasses, while black bears (*Ursus americanus*) and mountain lions used underpasses more frequently than overpasses (Clevenger and Waltho 2005). The location, type, and dimensions of a structure, as well as the habitat surrounding it, are all critical parameters that will determine use

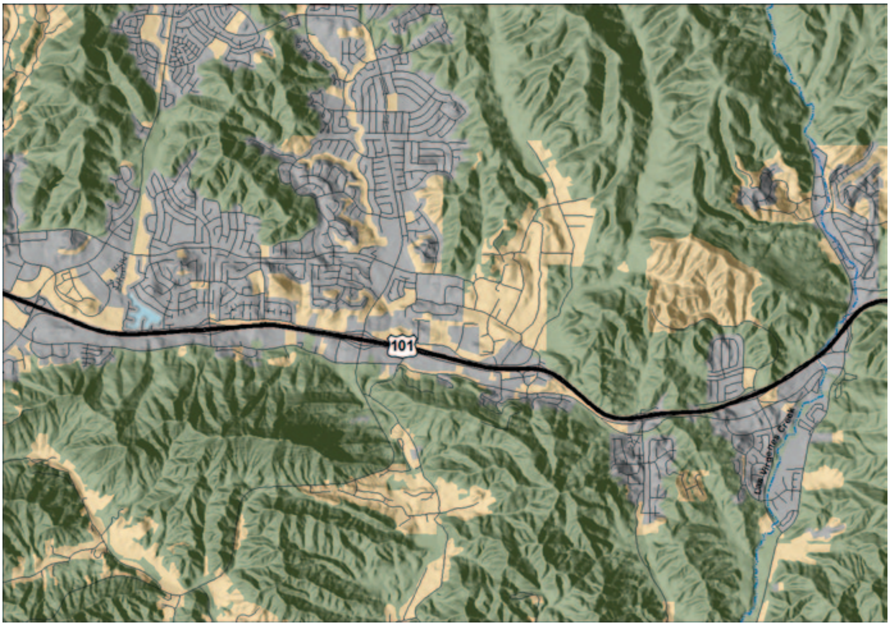

Fig. 15.3 Urbanization abuts much of US 101 such as along this 10 km stretch in Ventura and Los Angeles Counties, California, parallel to the Santa Monica Mountains to the south. Because the Freeway is also a development corridor, locations where natural habitat abuts the freeway are very limited. Moreover, the surrounding urbanization creates long narrow corridors perpendicular to the freeway before animals reach larger, more contiguous open space. For example, there is a culvert where Las Virgenes Creek (*thin blue line*, *right* side of image) crosses under the freeway, but the stream is surrounded by development for over a kilometer both north and south of US 101

by wildlife. For underpasses, variables including the length, width, and height of the structure and the presence of habitat on either side of the passage are important predictors of crossing for some species (Haas 2000; Forman et al. 2003; Ng et al. 2004). Ideally crossing structures of various types and sizes are needed at frequent intervals (Bissonette and Cramer 2008) to reduce impacts on the whole wildlife community. At a minimum, it is important to ensure that the crossing structure will be suitable for the species of greatest concern.

15.4.1.1 Overpasses: Types of Bridges for Wildlife Movement Over Roads

15.4.1.1.1 Wildlife Bridges

Wildlife bridges, also known as wildlife overpasses, are structures that are built over roads to allow animals to safely cross. They can range in width from 30 to 50 m on each end narrowing to 8–35 m in the center, to structures over 200 m wide (Jackson and Griffin 2000; Forman et al. 2003). Most wildlife bridges are vegetated and have a continuous strip of soil and native trees, shrubs, and grasses to provide suitable habitat for a variety of species. Wildlife overpasses are usually designed for large mammals (e.g., in Banff National Park, Canada, Clevenger and Waltho 2005), but if habitat is provided, small and medium-sized animals will use them as well (e.g., in the Netherlands, van Wieren and Worm 2001).

Overpass structures often incorporate specific habitat characteristics to attract wildlife. For example, some overpass structures have ponds on each end of the crossing to attract animals and provide habitat for amphibians. Overpasses may incorporate solid walls, earthen berms, dense vegetation, or a combination of these features on the outer edges (the “shoulders”) of the bridge to reduce light and noise disturbance for animals and block the view of the moving vehicles below. Small mammals, reptiles, and amphibians will use overpasses if cover is provided in the form of rock piles, logs, and bushes. Specific types of vegetation and strategic placement along a wildlife crossing can encourage crossings by birds and bats (Clevenger and Huijser 2010). Overpasses can even have a guiding-line function for birds to cross roads by following the natural habitat (Forman et al. 2003), as seen in Switzerland where woodland birds crossed a multilane highway using overpass structures (Keller and Pfister 1997).

Advantages of wildlife overpasses relative to crossing structures underneath roads include less confinement, less noise disturbance, and the maintenance of ambient conditions of rainfall, temperature, and light and therefore the ability to accommodate a wider variety of species and even to serve as habitat for smaller animals (Jackson and Griffin 2000). The major disadvantage is that they are usually the most expensive option (Glista et al. 2009). Thus, wildlife overpasses are uncommon, and as far as we know, none have yet been built in urban areas. This may be the result of high costs or because locations for them in urban areas are rare, as they require sufficient natural habitat on both sides of the freeway to accommodate the length and width of the structure. As more agencies and cities consider the importance of wildlife movement across roads, hopefully wildlife bridges will become an important feature in urban landscapes.

15.4.1.1.2 Road Bridges

Road bridges refer to bridges above a major linear element of infrastructure (e.g., a road or a railroad) that allow human access above it, generally via another road (van der Ree 2007). These structures are typically narrow, nonvegetated, and may contain a sidewalk for people. Very little attention has been paid to their potential utility for wildlife, perhaps not surprisingly since they are generally built to move cars, and they connect to roads on either end. However, these types of crossings are common in urban areas and may benefit wildlife in settings where options for connectivity are limited. Road bridges could be adapted for wildlife use by installing soil and vegetation along the edges of the bridge and creating a side path specifically for wildlife use. As with wildlife bridges (Clevenger and Huijser 2010), light and noise disturbance from vehicles may be reduced by using walls or vegetation.

As part of a widening project for the I-405 Freeway in Los Angeles, California, we have been monitoring the use of two existing bridges on I-405 where it passes through the Santa Monica Mountains (Fig. 15.2). We have documented coyotes, raccoons, rabbits, and striped skunks using the nonvegetated pedestrian sidewalks over I-405, and we have identified mule deer and bobcats on the sidewalks for these bridges as well, although they do not appear to have fully crossed the span (Schoonmaker and Riley 2011). As part of the mitigation for the freeway widening, the California Department of Transportation (Caltrans) is widening one of the bridges and adding a "wildlife sidewalk" that will be covered by decomposed granite and visually blocked from the freeway by a wall.

In another wildlife study in Los Angeles, remote cameras detected coyotes and mule deer using two road bridges over the US 101 Freeway as it passes through the Hollywood Hills (Boydston, Cooper, Ordeñana unpubl. data; Fig. 15.4). This same study also detected a mountain lion on the east side of US 101 in Griffith Park, a natural area surrounded by roads and home to the Hollywood sign (Chadwick 2013). Genetic analyses indicate that this animal came from the Santa Monica Mountains to the west (Riley et al. 2014), suggesting that he crossed both I-405 and US 101 Freeways to reach Griffith Park, potentially using road bridges (especially for US 101). If further research indicates that road bridges can assist with connectivity, we may see more designs for them to accommodate wildlife movements.

15.4.1.1.3 Canopy Bridges

Canopy crossings are another type of above-grade crossing structure designed exclusively for semiarboreal and arboreal species that use tree canopies for travel. They are designed to restore connectivity between forested habitats separated by major roadways, and their structure can vary based on the site, road type, road width, and focal species. Some crossings consist of thick ropes or cables that are anchored between trees or permanent fixtures such as signs or light posts (Beckmann et al. 2010). Rope bridge overpasses in northeast Queensland, Australia have been found to provide safe passage routes for arboreal rainforest mammals that are highly susceptible to road mortality (Goosem et al. 2006). In another study, rope bridges restored connectivity for arboreal species across a major highway in southeast Australia; complete crossings were recorded for five species, two of which were endangered, within 2 years of canopy crossing installation (Soanes and van der Ree 2009).

Again, there are no studies of canopy bridges in urban landscapes, as far as we know. However, in contrast to wildlife bridges, canopy bridges may be easy to implement in urban areas. There are many existing vertical structures along urban roads, including signs and lightposts, which could be used at minimal cost. There are also many street trees in cities, including in even the most intensely urbanized downtown areas, and many cities have movements to increase numbers

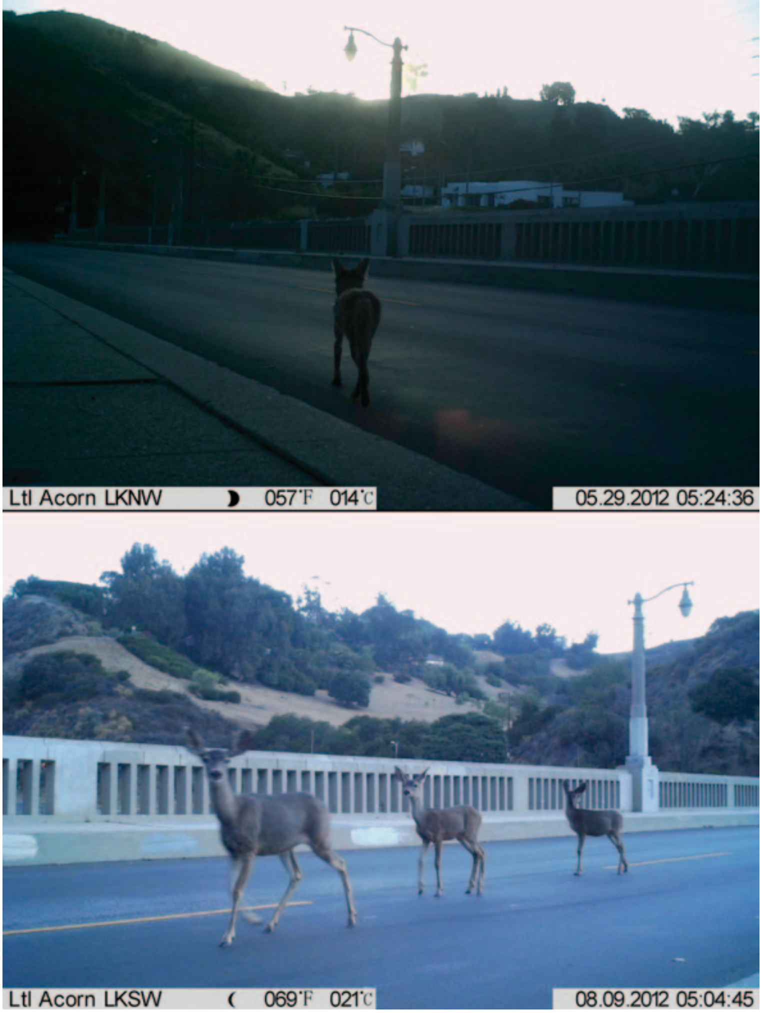

Fig. 15.4. Motion-activated cameras monitoring two bridges over US 101 in the Cahuenga Pass of Los Angeles, California detected occasional use by wildlife such as a coyote (*top* image) and a mule deer with fawns (*lower* image) (Boydston, Cooper, Ordeñana unpubl. data)

of street trees (e.g., the million trees project in New York City). When feasible (taking into consideration safety and maintenance concerns), street trees could be allowed to grow horizontally over roadways, or relatively short ropes or cables could be used to close the gaps, allowing arboreal species such as squirrels to safely cross. In regions where large deciduous trees are common along streets, this could be an effective strategy for reducing squirrel mortality on roads, which is often high especially in the fall when young animals are dispersing. Of course, canopy bridges are limited in that they are only useful for arboreal animals.

15.4.1.2 Underpasses: Structures Allowing Wildlife to Cross Under Roads

Wildlife underpasses include bridge underpasses, culverts, tunnels, and pipes, and their design and dimensions can vary considerably. As might be expected, larger animals require larger structures, although there can be some differences related to taxonomy and behavior as well (e.g., ungulates may be more reluctant to use tunnels than carnivores). Some underpasses have been built exclusively for wildlife use; for example, along Interstate 75 (Alligator Alley) in southern Florida, endangered Florida panthers (*P. concolor coryi*) and other species including bobcats, deer, raccoons, alligators (*Alligator mississippiensis*), and black bears regularly used underpasses specifically constructed for wildlife to safely cross under the highway (Foster and Humphrey 1995). Most tunnels used by wildlife were built for water drainage or human travel, but despite their original intent, these structures can be important linkages for wildlife (Forman et al. 2003; Ng et al. 2004). Moreover, underpasses that were not designed for animal passage can be modified to encourage wildlife use. For example, culverts with flowing water can be outfitted with a dry ledge for animal movement. In dry culverts, natural debris can be placed throughout larger underpasses to provide cover for smaller animals.

15.4.1.2.1 Bridge Underpasses

Bridges built where roads cross rivers, streams, and other roads form underpasses below them that range widely in size and are relatively common in urban areas. Usually built as features of a road rather than for wildlife, these underpasses can be quite valuable for movement by a variety of taxa because the bridges often span natural terrain features (Glista et al. 2009). Bridge underpasses can be adapted or modified (e.g., by adding vegetation) to further facilitate crossing by wildlife, including amphibians and semiaquatic and semiarboreal species (Beckmann et al. 2010).

Underpasses below large bridges typically provide plenty of air movement and light, as well as a sense of openness, which can increase the chances of crossing for many species. The height and width of bridge underpasses (the width of the road or stream, plus any side areas) make them viable even for large or more sensitive species such as ungulates. Bridges with open medians (i.e., where there is space between the lanes going in each direction) provide natural light in the middle which increases passage and can create small areas of natural habitat for small mammals, reptiles, and amphibians, but they are much louder than continuous bridges (Jackson and Griffin 2000). In our studies in the Los Angeles area, we have monitored a number of bridge underpasses along roads that have allowed movement under busy freeways, including I-405 and US 101, which would otherwise be impassable (Ng et al. 2004; Schoonmaker and Riley 2011). However, bridge underpasses in urban areas often cross channelized sections of rivers and streams, with concrete floors and tall, concrete walls, that are designed to accommodate heavy rainfall and may not be conducive for entrance or exit by wildlife.

15.4.1.2.2 Culverts

A culvert is a round or rectangular tunnel that allows water to flow underneath a road. They are usually made of concrete, smooth steel, or corrugated metal and are smaller than bridge underpasses. Depending on their specific characteristics, culverts can provide important avenues for animals to cross under roads and can increase connectivity in fragmented landscapes. Some species that have been shown to use culverts include black bear, mountain lion, wolf (*Canis lupus*), elk, deer, coyote, bobcat, raccoon, river otter (*Lontra canadensis*), nine-banded armadillo (*Dasypus novemcinctus*), Virginia opossum (*Didelphis virginiana*), marten (*Martes americana*), skunk, weasel (*Mustela sp*.), rabbit, squirrels, lizards, snakes, and amphibians (Yanes et al. 1995; Clevenger and Waltho 2000; Clevenger et al. 2001a; Cain et al. 2003; Ng et al. 2004; Dodd et al. 2004). Generally, culverts are larger than tunnels, although carnivores such as bobcats and coyotes will even utilize round pipe culverts that are only a meter in diameter (Alonso et al. 2014).

Concrete box culverts may be more effective than round metal culverts of the same size, because they provide more interior space for the same height and width, and because cement retains moisture and may be more like natural surfaces adjacent to the culvert (Ruediger and DiGiorgio 2007). Although the primary purpose of box culverts is to move water during flash floods and times of heavy rain, they can provide a corridor for many animals throughout the year. The installation of prefabricated box culverts under roads has proven successful in accommodating the movements of wildlife from large, wide-ranging mammals such as Florida panthers and black bears (Land and Lotz 1996; Evink 2002) to small, slow species such as spotted turtles (*Clemmys guttata*) (Kaye et al. 2005). In urban southern California we have recorded multiple species using even long (140 m) culverts, in one case as often as nearly every day for coyotes (Sikich and Riley 2012). Overall, culverts, even if not originally designed for wildlife, can be important for maintaining habitat connectivity across busy highways in an urban, fragmented landscape (Ng et al. 2004).

15.4.1.2.3 Tunnels

Smaller undercrossings such as “amphibian tunnels” and “wildlife pipes” ranging in size from 0.3 to 1.5 m (diameter or height) have been widely used in Europe to facilitate wildlife movement (Forman et al. 2003). These smaller undercrossings are particularly effective for amphibians, reptiles, and small mammals (e.g., Puky et al. 2007 in Hungary), and may be preferred to large structures by small mammals (McDonald and St Clair 2004). Small tunnels have also been effective for addressing fragmentation issues for sensitive species. In an Australian ski resort, the population structure and survival rates of mountain pygmy-possums (*Burramys parvus*) were restored using tunnels filled with rocks to safely cross a road bisecting a breeding area (Mansergh and Scotts 1989). Given the small size of wildlife tunnels, they are certainly amenable to use in urban landscapes. In places where important populations of amphibians or reptiles still exist in urban areas, these tunnels, accompanied by fencing, could be cost-effective solutions to connectivity and mortality concerns.

15.4.2 *Reducing Mortality*

Various mitigation techniques have been used to reduce WVCs. Most of these techniques can be separated into two categories: those aimed at modifying *human* behavior and those aimed at modifying *animal* behavior (Romin and Bissonette 1996; Forman et al. 2003; Beckmann et al. 2010).

15.4.2.1 Modifying Human Behavior

Public outreach and education to heighten community awareness about the impacts of roads on wildlife can be used to reduce road mortality, especially for sensitive species. We do not know of studies showing that public outreach has resulted in fewer WVCs, but it is still valuable for raising awareness about the problem and for generating public support for other mitigation measures (Beckmann et al. 2010). Warning signs are also regularly used to alert drivers to be watchful for wildlife on roads. However, Meyer (2006) found that standard deer warning signs had little effect in reducing WVCs, likely because drivers become habituated to them and any changed behavior (reduced speed or increased vigilance) disappears over time. Recent efforts have been made to improve warning sign technology, including interactive signs with sensors that detect and activate when large mammals are present near the highway. Night time speed reduction signs and variable message signs are also being used in some states as an alternative to traditional static signs (Ruediger and DiGiorgio 2007).

Temporary road closures are another mitigation measure that modifies human behavior. Such closures have been shown to enhance amphibian migration and reduce mortality during the breeding season (Forman and Alexander 1998). In a national park in Pennsylvania where amphibian mortality was common on roadways, road closures were put into place during the mass migration of five amphibian species and reduced mortality during this critical period (National Park Service 2006). Road closures, as well as people moving amphibians across roads in “bucket brigades” are rarely possible in urban landscapes where traffic is constant and the high demand for roads makes closures impossible. However, in Tilden Park, in Berkeley, California, a road through the park is closed for 5 months every year to protect newts migrating to and from wetlands to breed. This road is within the confines of the park, which makes the closure easier to manage, although even this park road is used by commuters, making the closure contentious.

Increasing the ability of drivers to see animals by adding lighting and removing vegetation (Putman 1997) may help reduce wildlife strikes along roads (Beckmann et al. 2010). Although some studies have shown reductions of large-mammal vehicle collisions by employing these techniques (Lavsund and Sandegren 1991), Beckmann et al. (2010) suggested these reductions may be a result of animals spending less time near the roadways because of the increased lighting and vegetation removal. In urban areas, there is often already considerable lighting along roads, so increasing it may not be an option, although roads through urban open space may be less well-lit, and increased roadway light can have other disadvantages for wildlife (see Sect. 15.2.3.1.4). Vegetation removal will also often occur near urban roads if there are residential or other with specific landscaping needs nearby.

15.4.2.2 Modifying Animal Behavior: Wildlife Fencing and Vegetation

Some mitigation methods directed at modifying animal behavior include reducing the attractiveness of the road or roadside for animals. One way this is accomplished is by removing carcasses from roadways so that carnivores and scavengers, both mammalian and avian, are not attracted to them. Planting unpalatable vegetation along roads may help reduce the attractiveness of roadsides to herbivores (Forman et al. 2003). Habitat alterations and maintaining natural vegetation in movement corridors leading up to a wildlife crossing structure are important to guide safe passage for many species. Vegetation enhances the effectiveness of the wildlife crossing by reducing the distance animals must travel between areas of natural habitat on either side of the road. It also provides cover and minimizes light and noise disturbance (Ruediger and DiGiorgio 2007). Restoring and vegetating the approaches to a wildlife crossing may be especially important in urban landscapes where human development may be close to the corridor on one or both sides, making its use by wildlife less likely (see Fig. 15.3). Although this can be hard to demonstrate conclusively, it is likely much more effective if natural corridors are wide as they lead away from the road to natural areas.

Wildlife-proof fencing is a common mitigation method used throughout the world to keep animals off roadways, ungulates in particular, because of concerns for motorist safety. These fences typically are 1.8 to 3.0 m high and consist of galvanized chain link or wire mesh fence material. Sometimes fences are buried into the ground approximately 0.6 m to keep animals from digging under them. A number of studies have found that wildlife-proof fencing can reduce ungulate–vehicle collisions (e.g., Ward 1982; Ludwig and Bremicker 1983; Dodd et al. 2007; McCollister and van Manen 2010). In Banff National Park, Clevenger et al. (2001b) reported an 80% reduction in ungulate–vehicle collisions after a wildlife-proof fence was constructed along the Trans-Canada highway. A species-specific fence was developed for European wildcats (*Felis silvestris*) along a motorway in southwestern Germany, which reduced road mortality by 83% (Klar et al. 2009). Tall fences can act as a barrier for medium-sized mammals as well as ungulates, and smaller mesh added to the bottom of a tall fence can be effective in keeping small mammals, amphibians, and reptiles from entering the roadway. Barrier walls along roads located in wetland systems can also be effective in reducing road mortality of reptiles and amphibians.

Where fencing is used for large mammals, escape ramps (jump-outs) or one-way gates will allow animals that become trapped inside the fencing, after finding their way over, under, or around the fence, to escape. For lower profile mammals such as badgers, small hinged doors at ground level can allow safe escape from the road ROW (Clevenger and Huijser 2010). In one study, earthen escape ramps were shown to be 6–12 times more effective than one-way gates in allowing deer to escape the ROW (Bissonette and Hammer 2000). Deer and elk most frequently use escape ramps, but bighorn sheep (*Ovis canadensis*), bears (*Ursus spp*.), moose, and mountain lions have been reported to use them as well (Clevenger and Huijser 2010).

There is no reason why wildlife-proof fencing will not also be effective in urban areas. It may be more difficult to install in urban settings, because there is often private land or residential or commercial areas abutting the road, making consistent application of the fencing difficult. For larger roads with a mandated ROW, road agencies may be able to dictate whether there is fencing and what kind it is. In California, state roads owned by Caltrans are mandated to have a fence along their ROW, although these fences are not always maintained, leading to holes or gaps. Another problem is that the ROW line may often veer far from the road, leading to significant areas of natural vegetation between the fence and the road, which is an attraction for wildlife. In one project in southern California, after replacing an old, unmaintained fence with a new wildlife-proof one, road mortality for coyotes was reduced by 88 % between the 3 years before the new fence and the 2 years after it (Sikich and Riley 2012).

15.4.2.3 Wildlife Fencing and Crossing Structures

The use of properly designed fencing (with wildlife escape mechanisms) or barrier walls in combination with carefully planned wildlife crossing structures will help reduce wildlife road mortalities and maintain habitat connectivity for a variety of species (Forman et al. 2003; Glista et al. 2009). This has been documented in many different places for large range of species, including in Australia for multiple marsupials, reptiles, and amphibians (Taylor and Goldingay 2003); in Florida both at Paynes Prairie for a range of vertebrates, especially snakes (Dodd et al. 2004), and on Big Pine Key with endangered Key deer (Braden et al. 2008); and in Arizona with elk (Dodd et al. 2007).

Further work is needed to understand how the dynamics of human-dominated landscapes influence wildlife use of crossing structures, but thoughtful application of this combination strategy is likely to be as effective in urban areas as elsewhere. In two studies in urban regions of southern California, we found that mitigation efforts, including clearing out existing underpasses, constructing wildlife-proof fencing, and installing one-way gates in one case (Sikich and Riley 2012), and adding wildlife fencing and modifying a system of existing culverts in another (Boydston and Crooks 2013), were effective in increasing culvert use and reducing road mortality for medium-sized mammals such as coyotes and bobcats.

15.5 Future Directions

Cities have large numbers of people moving across the landscape, and the transportation infrastructure associated with the massive daily migrations of cars runs counter to wildlife needs. Clearly further research is needed to understand: (1) wildlife responses to roads in urban environments; (2) how to modify existing roads; and (3) how to plan for urban expansion that will allow wildlife to also move through these highly fragmented landscapes. Some wildlife responses to roads may scale with road size, traffic volume, and the size of habitat fragments, whether the landscape is a city or a remote natural area. Other responses may be influenced by factors or interactions of factors that are unique to roads in cities, and identifying these interactions is an important step towards effective mitigation. Certain taxa have received more attention than others in road ecology research, with the bulk of the emphasis to date on carnivores and ungulates. While providing connectivity for these larger, wide-ranging mammals may serve a number of smaller animal species, much of the biodiversity in urban areas may require additional or other mitigation strategies along roads to persist. After conducting a meta-analysis using data from 75 studies, Rytwinski and Fahrig (2012) found that amphibians and reptiles, wide-ranging birds and large mammals, and species that do not avoid roads were particularly susceptible to road mortality.

To accommodate urban growth, cities frequently modify their transportation networks. These changes may often include widening of roads for greater carrying capacity of vehicles which can remove wildlife habitat. However, such changes also present opportunities to insert mitigation for wildlife, and in fact, improvements for wildlife that are incorporated into projects that have other goals (e.g., adding a carpool lane) may be relatively low-cost compared to a new wildlife bridge or other expensive mitigation.

Wildlife friendly improvement of roads in urban areas may require extensive funding, but improvements for wildlife may improve urban quality of life for humans beyond the ecosystem services of intact, connected natural areas. For example, planning for wildlife to safely cross roads may also help a city interested in creating friendlier passages for pedestrians and bicycles across freeways. While automotive demands will likely continue to increase for many years, the future of urban transportation planning may include an increased emphasis on public transportation networks such as rail lines and the incorporation of new technologies. Transportation alternatives that can move more people with less use of fossil fuels and lower emissions than cars may present new challenges for wildlife and make for even more complex equations in balancing environmental costs and benefits for wildlife and people. However, working towards improvements for wildlife now will increase the chances that wildlife can be part of the equation in the future.

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Chapter 16
Managing Aquatic Environments for Wildlife in Urban Areas

Steven J. Price, Joel W. Snodgrass and Michael E. Dorcas

Keywords Amphibians · Fish · Lakes · Lentic · Lotic · Reptiles · Riparian zones · Rivers · Streams · Wetlands

16.1 Introduction

Wetlands, streams, and riparian areas are often the center of wildlife conservation challenges in urban and suburban areas. Most aquatic environments and associated riparian zones exhibit high diversity and abundances of wildlife, yet these habitat types and the associated wildlife are among the most threatened by urbanization. In this chapter, we focus on the management of aquatic environments and their wildlife inhabitants in urban areas. Although a broad range of wildlife rely on urban aquatic environments, we focus on fishes, amphibians, and reptiles. Fishes, amphibians, and reptiles play important ecological roles (Godley 1980; Gilinsky 1984; Davic and Welsh 2004), exhibit high diversity and abundances in aquatic and riparian systems (Warren et al. 2000; Tuberville et al. 2005; Peterman et al. 2008), and often are useful in indicating the conditions of aquatic environments (Karr 1981; Welsh and Olliver 1998; Gibbons et al. 2000). We cover the following topics in this chapter: (1) the general importance of urban wetlands, streams, and riparian zones to wildlife; (2) aquatic habitat types that occur in urban areas; (3) the effects of urban areas and urbanization on local and regional populations of fishes, amphibians, and semiaquatic reptiles; (4) the critical elements necessary for effective management of aquatic environments for fishes, amphibians, and reptiles in urban and suburban areas.

S. J. Price (✉)
Department of Forestry, University of Kentucky,
Thomas Poe Cooper, Lexington, KY 40546-0073, USA
e-mail: steven.price@uky.edu

J. W. Snodgrass
Department of Biological Sciences, Towson University, Towson, MD 21252, USA

M. E. Dorcas
Department of Biology, Davidson College, Davidson, NC 28035-7118, USA

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16.2 The Importance of Urban Aquatic Environments for Wildlife

Freshwater systems and associated riparian communities make up a small percentage of the earth's surface, yet these environments are critical for many groups of wildlife and often exhibit high levels of productivity and species richness (Petranka and Murray 2001; Brinson and Inés Malvárez 2002; Gibbons et al. 2006). Fishes, amphibians, and reptiles are the dominant vertebrate groups in aquatic systems, reaching high population densities and biomass (Godley 1980; Gilinsky 1984; Petranka and Murray 2001; Gibbons et al. 2006; Peterman et al. 2008). The maintenance of preferable abiotic and biotic conditions within and adjacent to aquatic environments is necessary for the persistence of aquatic and semiaquatic wildlife. Urbanization often results in the destruction, degradation, and fragmentation of habitat, which collectivity represents a major threat to fishes, amphibians, and reptiles (Wang et al. 2001; Baillie et al. 2004; Cushman 2006; Hamer and McDonnell 2008, 2010). Additionally, because vast quantities of water are required for the proper functioning of an urban area (Wolman 1965; Kennedy et al. 2007), significant alterations to the water cycle, reductions in water supplies, and chemical contaminants stress the freshwater ecosystems in urban areas (Chap. 4, Fitzhugh and Richter 2004).

16.3 Aquatic Habitat Types in Urban Environments

16.3.1 Naturally Occurring Aquatic Habitats

Many natural aquatic habitats are destroyed during the urbanization process; however, some persist, particularly larger aquatic systems involved in draining runoff. Large rivers historically attracted development and, although modified greatly by humans, persist as significant elements in many modern cities (Grischek et al. 2002). Thus, aquatic and semiaquatic wildlife, particularly fish and reptiles, which occupy large, riverine systems sometimes are present in urban areas (Conner et al. 2005; Meador et al. 2005; Barrett and Guyer 2008). Alternatively, smaller streams, especially ephemeral and intermittent streams, can be destroyed or lost due to changes in hydrology or burial (i.e., directed into underground pipes or other drainage structures, or completely paved over). In Ohio, Roy et al. (2009) estimated that urbanization resulted in a loss of 93 and 46 % of ephemeral and intermittent stream length, respectively. In Baltimore, up to 70 % of the stream length of smaller watersheds was buried as a result of urbanization (Elmore and Kaushal 2008). Salamander, anuran, and fish populations associated with these low-order streams are often negatively impacted by urbanization (Wang et al. 2001; Barrett and Guyer 2008; Price et al. 2011).

Wetlands and lakes also persist in urban landscapes, but the destruction of wetlands outpaces that of stream systems (Ehrenfeld 2000). For wetlands, drainage and filling often preceded urbanization when lands were converted for agricultural uses (Biebighauser 2007). Urbanization often leads to further losses. For example, in Pennsylvania, urbanization reduced natural wetland density by over 50 % from approximately 15 % of the land cover to 7 % (Rubbo and Kiesecker 2005). The dominant vertebrate taxa in wetlands are amphibians and reptiles, and as wetlands become altered or destroyed from urbanization, species often disappear (Gibbs 1993; Guzy et al. 2012). Because of their water storage capacity and aesthetic appeal, most natural lakes, formed by geological processes, persist in urban areas. However, the shores of many urban lakes have been extensively developed and modified, negatively affecting populations of fish, reptiles, and amphibians (Jennings et al. 1999; Woodford and Meyer 2003).

16.3.2 Human-Created Aquatic Habitats

As part of the urbanization process, environments that at least superficially resemble natural wetlands, lakes, or streams are often created. Human-created environments range from small garden ponds designed to attract wildlife (Beebee 1979) to stormwater management structures (collectively referred to here as stormwater ponds) and reservoirs. The primary function of stormwater ponds is protection of water quality and hydrological processes in natural wetlands and streams that receive runoff from impervious surfaces (Villareal et al. 2004). Stormwater ponds may mitigate the loss of natural wetlands and act as habitat for aquatic and semiaquatic wildlife (Stahre and Urbonas 1990), but habitats within these artificial ponds typically are of lower quality than natural wetlands and may contain chemicals that are toxic to wildlife (Bishop et al. 2000a, b). Ultimately, the value of stormwater ponds as habitat for wildlife will depend on the amount of pollution they accumulate, their hydroperiod, and the availability of and proximity to natural aquatic systems (Gallagher et al. 2014; Birx-Raybuck et al. 2010; Brand and Snodgrass 2010).

16.3.3 Riparian Zones and Adjacent Terrestrial Environments

Riparian zones, generally defined as an area of interface between aquatic systems and adjacent terrestrial systems (Naiman et al. 2005), often persist along streams in urban areas. These zones may be required by local laws to protect water quality as part of stormwater management practices or as green spaces with aesthetic value, or both. Although riparian zones serve water quality protection functions in urban systems (Gilliam 1994; Correll 1997), they also may serve directly as habitat for a range of aquatic and semiaquatic species (Ehrenfeld and Stander 2010). Other remaining patches of undeveloped open space and landscaped upland areas adjacent to aquatic systems may serve as habitat for semiaquatic wildlife as they move among wetlands to complete their life cycles or disperse. These patches may take the form of green spaces, parks, roadsides, golf courses, and maintained gardens around residential, commercial, and industrial buildings.

16.4 Urban Impacts on SemiAquatic and Aquatic Wildlife

16.4.1 The Effects of Urbanization on Patterns of Distribution, Abundance, and Species Richness

Research suggests an exponential decline in richness of fish, amphibian, and reptile species with increasing urbanization (Klein 1979; Wang et al. 2000; Spinks et al. 2003; Hamer and McDonald 2008, 2010), and even low-intensity development can reduce richness and abundance (Weaver and Garman 1994; Kemp and Spotila 1997; Willson and Dorcas 2003; Price et al. 2013). Studies by Price et al. (2011, 2012) suggest that some amphibian populations decline rapidly with the conversion of forested land to urban land, although a significant time lag may occur between population declines and urbanization, especially for longer-lived aquatic and semiaquatic wildlife species (Findlay and Bourdages 2000; Eskew et al. 2010a, b).

Conversely, some aquatic and semiaquatic wildlife species may not be as sensitive to urbanization. Native fishes, amphibians, and reptiles often persist in urbanized aquatic habitats, particularly under the right set of conditions (Conner et al. 2005; Riley et al. 2005; Rubbo and Kiesecker 2005; Barrett and Guyer 2008; Leidy et al. 2011), and may have abundances equal or greater than populations in rural areas (Klein 1979; Fraker et al. 2002; Price et al. 2013). As with other groups of organisms, declines in native species richness can sometimes be offset by introduction of nonnative species in urban areas (Meador et al. 2005). However, it appears that urbanization results in the persistence of a few relatively tolerant and widespread native species (i.e., urban exploiters and urban adapters), extirpation of relatively intolerant, more narrowly distributed species (i.e., urban avoiders), and the introduction of already widespread nonnatives (Chap. 7).

16.4.2 Mechanisms Responsible for Patterns

16.4.2.1 The Effects of Urban Hydrology

Urbanization results in increased water level fluctuations in natural lakes, rivers, wetlands, and streams as well as in human-created habitats such as stormwater ponds (Reinelt and Taylor 2000; Coops et al. 2003; Kentula et al. 2004; Ostergaard et al. 2008; Wantzen et al. 2008). Urban aquatic systems show short-term fluctuations with individual storm events (Hirsch et al. 1990) and longer-term changes in hydroperiod (Barringer et al. 1994; Paul and Meyer 2001; Schoonover et al. 2006). These fluctuations are caused by loss of vegetation and associated evapotranspiration, increase in impervious surfaces that increase storm runoff (including sediment) directly to aquatic habitats, and reduced groundwater recharge and ground water tables found in urban systems (Barringer et al. 1994; Pizzuto et al. 2000; Paul and Meyer 2001). Collectively, these factors can dramatically alter the geomorphology of aquatic systems in urban areas (Wolman 1967; Arnold et al. 1982; Gregory et al. 1994; Booth and Jackson 1997, see Chap. 4).

Modified hydrologic regimes affect populations of fishes, amphibians, and reptiles in a variety of ways. For fish and amphibians in stream systems, increased peak flow events and lower base flow conditions combined with loss of in-stream habitat due to sedimentation lead to decreased population densities (Orser and Shure 1972; Bain et al. 1988; Miller et al. 2007; Barrett et al. 2010). For example, Barrett et al. (2010) found that larval two-lined salamanders (*Eurycea cirrigera*) on substrates typical of urban streams (i.e., sand) were flushed downstream at significantly lower water velocities than larva on rock-based substrates, suggesting that the synergistic effect of water flow and substrate modification reduces larval survivorship in urban areas. However, low base flow conditions may also strongly influence populations. Low abundances of two-lined salamander larva in sediment-choked urban streams were due, in part, to their inability to migrate to hyporheic zones during periods of low flow (Miller et al. 2007). Low flow combined with the accumulation of fine sediments in urban streams also play a significant role in degrading urban stream fish assemblages, resulting in the loss of lithophilic spawners from urbanized streams (Wang et al. 2001; Helms et al. 2005). Conversely, urban hydrology also may lead to the widening and deepening of streams, especially when drainage is highly modified due to development. In western Georgia, Barrett and Guyer (2008) documented greater reptile species richness in urban watersheds than rural watersheds, and suggested that the widening of streams promoted species associated with larger, open canopy streams and rivers.

Urban wetlands also exhibit modified hydrologic regimes. Ephemeral wetlands often are converted, either intentionally or unintentionally, to permanent wetlands or ponds in urban areas. This phenomenon can lead to the establishment of fish (Kentula et al. 2004), reptile (Barrett and Guyer 2008), and invertebrate populations (Riley et al. 2005) typically not present in ephemeral wetlands. Many amphibian species are negatively impacted by the introduction of fish and some invertebrates; only those species that have anti-predatory behaviors or are unpalatable, such a bullfrogs (*Lithobates catesbeianus*), appear to have high survival in permanent urban water bodies, whereas species that are palatable to fish and invertebrates usually do not persist (Rubbo and Kiesecker 2005). Similarly, some species, such as bog turtles (*Glyptemys muhlenbergii*), that inhabit shallow wetlands have experienced population declines when urban development leads to increased discharges of stormwater runoff into wetlands (Torok 1994). The conversion of ephemeral wetlands to permanent wetlands or ponds may promote local fish diversity, although increases in diversity often result from the introduction of relatively tolerant and widespread native species or widely introduced nonnatives species (Brown et al. 2009). Conversely, increased ground water withdrawal in urban areas may result in rapid drying of some aquatic habitats, affecting survival of larval amphibians (Bunnell and Ciraolo 2010; Guzy et al. 2012) and potentially leading to the decline of semiaquatic reptile populations that feed on amphibians.

16.4.2.2 The Effects of Urban Pollution

In urban areas, a broad range of pollutants may accumulate within aquatic environments, which can have lethal and sublethal effects on wildlife (Chap. 10). Weber and Bannerman (2004) exposed fathead minnows (*Pimephales promelas*) to urban stream water and recorded reduced fecundity, breeding activity, and development of secondary sexual characteristics among males, suggesting at least a sublethal role of pollutants and water quality in reducing or eliminating fishes from urban streams. Increased levels of metals (i.e., zinc, lead, etc.), nitrogen, and sediment in urban aquatic habitats have been shown to cause mass mortality in wood frogs (*Lithobates sylvaticus*) (Snodgrass et al. 2008), and reduce growth, survivorship, and development rates in a variety of amphibian species (Boone and Bridges 2003; Carey et al. 2003). The accumulation of pollutants in the tissues of aquatic wildlife from urban systems is also suggestive of a role for pollutants in degrading urban fish assemblages (Ney and Van Hassel 1983; Campbell 1994), and may lead to significant genetic and developmental abnormalities for species with long-life spans such as turtles **(**Crews et al. 1995; Lamb et al. 1995). For example, common snapping turtle (*Chelydra serpentina*) populations often have high levels of contaminants, especially polychlorinated biphenyls (PCBs), in their fat (Helwig and Hora 1983) and eggs (de Solla et al. 2001), and contamination levels are positively correlated with proximity to industrial urban areas (Ashpole et al. 2004). Additionally, high levels of PCBs have an estrogenic effect resulting in alteration of sex differentiation in turtles (Bergeron et al. 1994). Finally, increased levels of synthetic estrogens are often associated with urban aquatic environments due to human use of birth control; Skelly et al. (2010) indicates that high levels of synthetic estrogens in urban ponds and wetlands may be responsible for sexual abnormalities (i.e., testicular oocytes) in male green frogs (*Lithobates clamitans*).

Urbanization also can lead to increases in conductivity of streams (Paul and Meyer 2001) and wetlands (Glooschenko et al. 1992). Several factors contribute to increased conductivity; the most problematic of which are the salts placed on roads as deicing agents (e.g., NaCl, MgCl, and CaCl; Van meter et al. 2011). Road salts readily dissolve in surface and ground waters resulting in seasonal or year round elevations of ion concentrations (Novotny et al. 2008; Gallagher et al. 2014). Road salts can reduce the abundance and species richness of macroinvertebrates (Demers 1992; Bridgeman et al. 2000), an important food source for fish, amphibians, and some reptiles. Salts applied to roads also affect osmoregulation in amphibians (Shoemaker and Nagy 1977), and reduce embryonic and larval survival of wetland-inhabiting amphibians at moderate (500 µS) and high (3000 µS) conductivities (Karraker et al. 2008).

Eutrophication, an increase in nutrients, has long been recognized as a problem for lentic systems in many parts of the world (Schindler 1978) and can be associated with urbanization (Moore et al. 2003). Eutrophication is dependent on extent and type of urban development, behaviors of humans within the catchment, presence of wastewater treatment facilities (WTFs), and extent of storm water drainage (Paul and Meyer 2001). Additionally, leaking sewer systems, illicit discharges, improperly functioning septic tanks, and nonpoint sources (e.g., fertilizer application) can contribute to eutrophication in urban streams (Adams and Lindsey 2010). Eutrophication can decrease dissolved oxygen levels causing problems for many susceptible amphibians (Mills and Barnhart 1999; Werner and Glennemeier 1999; Woods et al. 2010) and can reduce or eliminate fish eggs and larvae (Limburg and Schmidt 1990). Despite the fact that eutrophication may enhance populations of semiaquatic turtles through the stimulation of aquatic plant growth, a food source for numerous turtle species (Knight and Gibbons 1968), high levels of nutrients also enhance populations of ecto- and endoparasites. Brites and Ratin (2004) noted that semiaquatic turtles (i.e., *Phrynops geoffroanus*) had greater rates of leech and hemogregarine parasitism in urban areas compared with agricultural areas.

16.4.2.3 The Effects of Introduced Species, Human Subsidized Species, and Human Interactions

Numerous nonnative species have been introduced, either intentionally or unintentionally, to urban areas. Additionally, some native species have obtained considerable population sizes in urban areas as a result of introductions or subsidies from human populations (McKinney 2002, 2008). Many introduced and human subsidized species have the ability, through habitat modification, predation, and/or competition, to reduce populations of native aquatic and semiaquatic species in urban areas. Furthermore, interactions with humans can negatively impact populations of some native wildlife.

Nonnative and invasive aquatic plants have been introduced to urban areas throughout the world (Arthington et al. 1983; Pauchard et al. 2006; Seilheimer et al. 2007), dramatically altering aquatic environmental conditions. For example, an invasive genotype of common reed (*Phragmites australis*) has become a dominant species in many coastal wetlands of the USA, especially where urban and suburban development is adjacent to wetlands (King et al. 2007). The common reed affects the hydrology, hydroperiod, and drainage density of a marsh, and negatively impacts habitat for fishes (Weinstein and Balletto 1999). Indeed, fewer juvenile fish occur in marshes where common reed is dominant (Able et al. 2003; Raichel et al. 2003; Osgood et al. 2003) compared to marshes dominated by native cordgrass (*Spartina alterniflora*). Additionally, Zedler and Kercher (2010) suggested that because common reed reduces the topographic heterogeneity and raises the marsh plain elevation, the number and area of isolated pools within the marsh is reduced, which could negatively affect some amphibian and reptile populations (Meyerson et al. 2000).

Numerous nonnative and invasive animals are introduced or stocked into urban aquatic environments in the USA. Fish are commonly stocked in urban ponds; species include largemouth bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), green sunfish (*Lepomis cyanellus*), yellow bullhead (*Ameiurus natalis*), common carp (*Cyprinus carpio*), and western mosquitofish (*Gambusia affinis*; Brown et al. 2009). Such introductions can negatively affect populations of native fish (Weber and Brown 2011), and are especially detrimental to amphibian populations due to fish predation on amphibian larva and adults (Rubbo and Kiesecker 2005). Similarly, bullfrogs, a species native to eastern North America, have been introduced extensively in urban areas in over 40 countries and 4 continents (Lever 2003; Ficetola et al. 2010). Bullfrogs outcompete and depredate native amphibian species (Blaustein and Kiesecker 2002) and can spread diseases (Kiesecker et al. 2001). Several introductions of aquatic and semiaquatic animals have resulted from the release of unwanted pets; the most notorious being the release of red-eared sliders (*Trachemys scripta*) in urban areas of western North America, Europe, Asia, and Australia (Bury 2008; Moll 1995). Sliders outcompete European turtles (*Emys orbicularis*) for preferred basking sites (Cadi and Joly 2003), negatively affect survival (Cadi and Joly 2004), and may compete with native turtle species for food and nesting sites.

Urban terrestrial environments also may present challenges to the survival of semiaquatic wildlife because of the introduction and/or subsidization of predators (Prange et al. 2004). Subsidization of predators occurs when humans alter resources to increase the density of the predator above levels that would occur without the human-introduced resources (Gompper and Vanak 2008). Raccoons (*Procyon lotor*), striped skunks (*Mephetis mephitis*), coyotes (*Canis latrans*), Virginia opossums (*Didelphis virginiana*), common ravens (*Corvus corax*), feral cats (*Felis silvestris*) and dogs (*Canis lupus familiaris*) can attain large populations in urban areas due to human subsidies (Churcher and Lawton 1987; Crooks and Soulé 1999; Boarman et al. 2006). Predation by human-subsidized predators can limit recruitment and result in declines of turtle populations (Burke et al. 2005; Strickland et al. 2010). Turtles restricted to nesting in small patches of habitat, often found around urban ponds, may experience greater rates of nest depredation than in rural settings (Marchand et al. 2002; but see Foley et al. 2012).

Increased presence of humans in urban environments increases the possibility of persecution, disturbance, and collecting by humans. Human persecution of snakes is well-documented and many snakes are killed on sight. Watersnakes (*Nerodia*) are often mistaken as venomous (and potentially dangerous) cottonmouths (*Agkistrodon piscivorus*) and killed around aquatic habitats (Gibbons and Dorcas 2004). Likewise, snapping turtles may be particularly vulnerable to persecution because of their perceived aggressiveness when found on land; in many cases they are killed and occasionally consumed (Ernst and Lovich 2009). Some species (e.g., wood turtles, *Glyptemys insculpta*) are unsustainably collected by humans in suburban parks (Garber and Burger 1995). Other wildlife (e.g., anurans) may be indirectly affected by increased human presence. Traffic noise has been shown to mask anuran advertisement calls (Bee and Swanson 2007), reduce calling intensity (Legange 2008), and disorientate individuals (Barber et al. 2010), collectively making it more difficult for female anurans to locate male anurans at urban breeding sites.

16.4.2.4 The Effects of Shoreline and Riparian Development on Wildlife in Urban Areas

The development of the shorelines of streams, lakes, and wetlands in urbanized watersheds degrades habitat and affects terrestrial–aquatic linkages. Development of shorelines severs the linkages between terrestrial and aquatic ecosystems, leading to the loss or reduction of detritus in near-shore sediments (Paul and Meyer 2001; Francis et al. 2007; Roberts and Bilby 2009), macrophytes (Jennings et al. 2003), coarse woody debris (Christensen et al. 1996; Finkenbine et al. 2000; Francis and Schindler 2006), and terrestrial insect subsidies. Shoreline engineering further degrades or destroys littoral habitat (Sukopp 1971; Radomski and Goeman 2001; Elias and Meyer 2003), ultimately leading to decoupling of benthic-pelagic food webs (Francis and Schindler 2009).

Riparian and near-shore vegetation represent critical habitats for aquatic and semiaquatic wildlife (May et al. 1997; Reese and Welsch 1998; Woodford and Meyer 2003). Development of shoreline and riparian zones result in reduced fish growth and health (Eitzmann and Paukert 2009), with species of recreational interest, such as largemouth bass affected most (Francis and Schindler 2009; Doi et al. 2010). Shoreline development also leads to reduced amphibian abundances (Woodford and Meyer 2003). These effects are likely due to both a reduction in habitat used for foraging (May et al. 1997) and change in diets induced by the decoupling of aquatic–terrestrial linkages (Sass et al. 2006; Francis and Schindler 2009).

Introduction of human structures to shoreline and aquatic environments, such as culverts, affects riparian and near shore areas, and may reduce movement of wildlife and fragment populations. Even small structures, such as box culverts, can reduce upstream movements of small fishes and modify the in-stream environment (Beasley and Hightower 2000; Bouska and Paukert 2009). Larger structures such as dams, which often provide hydroelectric power to urban areas, can result in loss of genetic diversity and reduce species occupancy and abundance (O'Hanley and Tomberlin 2005; Sheer and Steel 2006; Eskew et al. 2012; Roberts 2012; Hunt et al. 2013).

Inputs of large woody debris are reduced in urban aquatic environments (Elosegi and Johnson 2003; Spinks et al. 2003). Basking is an important thermoregulatory behavior of semiaquatic reptiles, and several studies have documented a positive relationship between basking sites or deadwood (i.e., logs) and semiaquatic reptile abundance (DonnerWright et al. 1999; Lindeman 1999; Reese and Welsch 1998). Thus, the removal of deadwood and other potential basking sites may negatively affect reptile populations. Yet, even if basking sites remain, increased human presence in and around the aquatic environment may limit basking opportunities or cause abandonment of basking sites (Moore and Seigel 2006).

16.4.2.5 The Effects of Development in Terrestrial Environments on Wildlife in Urban Areas

Most semiaquatic wildlife species depend on surrounding terrestrial environments for various life-history functions (Semlitsch and Bodie 2003; Rowe et al. 2005; Bowne et al. 2006; Roe et al. 2006; Steen et al. 2006; Harden et al. 2009)**.** At the landscape-level, amphibians and reptiles often are distributed as a series of localized populations centered on aquatic environments and connected via migration (i.e., metapopulations, see Gill 1978; Marsh and Trenham 2001; Dodd and Smith 2003; Smith and Green 2005). Thus, the extent of urbanization surrounding aquatic environments may strongly influence population persistence (e.g., Knutson et al. 1999; Spinks et al. 2003; Price et al. 2005). Furthermore, urbanization often reduces the density of aquatic habitats (Rubbo and Kiesecker 2005), which increases the distance between suitable aquatic sites and affects recolonization, which is often critical for the maintenance of populations across landscapes (Semlitsch and Bodie 1998).

Several studies have shown a negative relationship between amphibian occupancy or abundance and amount of land in urban or suburban cover at large-spatial scales (see Hamer and McDonnell 2008). For example, Rubbo and Kiesecker (2005) detected few occurrences of the forest-dependent wood frog (*Lithobates sylvaticus*) and spotted salamander (*Ambystoma maculatum*) in urban wetlands compared to wetlands surrounded by forested land. Willson and Dorcas (2003), studying salamanders in a suburban landscape in North Carolina, USA, showed that the abundance of stream-dwelling salamanders was highly correlated with the amount of undisturbed land within the entire stream catchment, but was not correlated with the amount of undisturbed land within required buffer zones.

Many amphibians and reptiles migrate to terrestrial environments to nest, forage, hibernate, or disperse to adjacent aquatic environments. Urban wetlands and streams often are surrounded by roads, a cover type either behaviorally avoided or a potential source of significant mortality from vehicular traffic (Gibbs 1998; Mazerolle 2004; Steen and Gibbs 2004; Mazerolle et al. 2005; Andrews et al. 2008). For species, such as turtles, that require high adult survival to maintain viable populations (Congdon et al. 1993), mortality during terrestrial movements may represent a significant threat to their long-term persistence in urban areas. Pittman et al. (2011) estimated annual survival of a suburban bog turtle population to be 0.89, a rate likely lower than required to maintain a stable population. Eskew et al. (2010a) found annual survival of mud turtles (*Kinosternon subrubrum*), a species known for extensive terrestrial movements (Harden et al. 2009), to be lower in a suburban environment than estimates from rural environments. Thus, it appears that fragmentation due to roads and other anthropogenic surfaces in urban areas may serve to isolate populations by hindering critical metapopulation processes. Indeed**,** genetic divergence among amphibian populations is positively correlated with urban development in the surrounding landscape (Reh and Seitz 1990; Hitchings and Beebee 1997; Safner et al. 2011); however, for long-lived semiaquatic reptiles, significant time lags between urban development and changes to genetic structure likely occur (Pittman et al. 2011).

As mentioned above, some species of aquatic and semiaquatic wildlife persist in urban areas. Barrett and Guyer (2008) determined that the alteration of streams from semipermanent, closed-canopy systems to open vegetation and deeper, warmer water favored some riverine turtles and snakes. Specifically, Barrett and Guyer (2008) suggested that urbanization may not be as detrimental to reptiles as amphibians because reptiles are able to recolonize urban areas more easily and their skin and amniotic eggs are less affected by changes in water quality. Furthermore, urbanization has led to gains in some types of aquatic habitats (Dahl 2006, 2011), especially permanent ponds often inhabited by semiaquatic reptiles, fish, and some amphibians (e.g., bullfrogs). From 1998 to 2004, over 280,000 ha of ponds were created in the lower 48 USA, due, in part, to the construction of stormwater detention ponds, ponds in suburban parks, and ponds on recreational lands, such as golf courses (Tilton 1995; Dahl 2006). In particular, golf course ponds have been shown to provide suitable habitat for semiaquatic reptiles and some amphibians in urban areas (McDonough and Paton 2006; Harden et al. 2009; Foley et al. 2012; Puglis and Boone 2012; Guzy et al. 2013; Price et al. 2013).

16.5 Elements of Effective Management of Aquatic and Semiaquatic Animals in Urban Aquatic Habitats

Effective management strategies that benefit multiple populations and species are built on identification of key stressors and development of tools that mitigate their sources (Wenger et al. 2009). Because stormwater runoff is widely recognized as the most significant stressor to urban aquatic systems (Walsh et al. 2005), we begin this section with a description of stormwater management techniques and then move to habitat restoration, reintroduction, and translocation, and habitat protection and planning. We caution that these general strategies, and may not be suitable for every given species; managing individual species in urban and suburban regions requires a detailed knowledge of life history, which is not always available, even for relatively common species.

16.5.1 Stormwater Management

Improving stormwater management facilities and modifying human behavior near aquatic habitats can reduce the impacts to wildlife associated with runoff. A variety of control measures can be used to slow, retain, and absorb pollutants and excess water associated with stormwater runoff (Tsihrintzis and Hamid 1997). First, in most developed countries, water from urban communities is often treated via water treatment facilities (WTFs) prior to release into the environment. This has obvious positive effects on fish, amphibians, and reptiles (and numerous other taxa) as the wastes removed include pollutants such as plastic bags, condoms, fecal matter, toilet paper, and colloidal and dissolved organic matter (i.e., bacteria, urine and soaps; Adams and Lindsey 2010). However, leaking sewage pipes associated with dated sanitary sewer infrastructure and sewage overflows associated with systems too small for the demands placed on them can be significant sources of contaminated water and nutrients to urban lakes and streams, and efforts to modernize sewage systems are needed in many larger cities.

Best management practices (BMPs), including both structural and nonstructural measures, should also be used near urban aquatic habitats. Structural control measures are physical structures that collect and treat runoff that does not go to WTFs. For example, the placement of stormwater ponds adjacent to streams and wetlands prevents chemical contamination, sedimentation, and variability of water flow (Tsihrintzis and Hamid 1997; Behera et al. 1999; Harrell and Ranjithan 2003), benefitting fish, amphibian, and reptile populations. In addition to reducing pollutant loading and excess water, stormwater ponds also may provide habitat for some amphibian and reptile species (Simon et al. 2009; Ackley and Meylan 2010; Birx-Raybuck et al. 2010; Brand and Snodgrass 2010; Hamer et al. 2012; Le Viol et al. 2012), although the high levels of pollutants in runoff may affect survival and reproduction (Snodgrass et al. 2008). Therefore, BMP structures that are expected to accumulate large amounts of pollutants should be managed in ways that discourage wildlife use (e.g., vegetation kept to a minimum). Nonstructural measures include public education, street cleaning, reducing fertilizer application and zoning to restrict population densities near waterways (Tsihrintzis and Hamid 1997). Effectively managing stormwater and runoff should involve a combination of WTFs, structural control measures, and nonstructural control measures; together they can lead to the reduction of chemicals and other pollutants near wetlands and streams.

16.5.2 Habitat Restoration

A goal of habitat restoration is to support a wide variety of native species and maximize resilience and persistence of populations to environmental disturbances (Miller and Hobbs 2007). Restoring habitat suitable for aquatic and semiaquatic wildlife in urban environments varies among the aquatic environments found in urban areas. The restoration of wetland habitat relies on the restoration of appropriate hydroperiod, which often leads to a decline in populations of introduced, predatory fish that prey upon amphibians and native fishes (Semlitsch 2000). Restoring stream habitat is also related to management of stormwater. The removal of stormwater pipes that directly connect impervious surfaces to streams and lakes limits frequent excessive flows (Walsh et al. 2005), which negatively affect salamanders (Barrett et al. 2010) and fishes (Bain et al. 1988). In drier landscapes, restoration of stream flows and natural disturbance regimes may reduce populations of nonnatives that lack the adaptations to cope with flow disturbances and assure wetted habitats are available on the appropriate seasonal basis to support the life cycles of native species (Marchetti and Moyle 2001; Harvey et al. 2006; Bradford and Heinonen 2008). Other stream restoration techniques include bank stabilization and provisioning of instream structural complexity. These techniques are believed to reduce sediment loads in critical riffle habitats and provide smaller animals with hiding places from predators (Roni et al. 2005; Bernhardt and Palmer 2007).

Dredging and removal of contaminated sediment in combination with elimination or reduction of point and nonpoint nutrient inputs can reverse eutrophic conditions in urban lakes (Ruley and Rusch 2002), and increasing the piscivorous to planktivorous fish ratio can be used to shift lake trophic states and promote the establishment of littoral zone vegetation (Jeppesen et al. 1990). Restoration of nearshore areas may involve adding coarse woody debris and restoring native macrophyte communities. Yet, the addition of course woody debris alone may not reverse the effects of shoreline urbanization on fish populations, at least in the short-term (Sass et al. 2012).

Efforts to revegetate riparian zones and terrestrial environments surrounding ponds and streams can reduce excessive flows and improve water quality; additionally it will provide amphibians and reptiles with the critical upland conditions necessary to complete their life cycles (Semlitsch and Bodie 2003; Crawford and Semlitsch 2007). Even leaving a buffer of unmowed grass around wetlands has been shown to positively affect local amphibian and turtle populations on golf courses (Foley et al. 2012; Puglis and Boone 2012). Revegetation of riparian zones also will likely benefit fish populations by supporting insect populations, increasing leaf litter inputs, and adding large woody debris to aquatic environments. In turn, semiaquatic reptiles (e.g., watersnakes) may benefit from the increased abundance of fish prey.

Efforts to restore habitat for aquatic wildlife in urban areas should consider impacts at the landscape scale (Brooks et al. 2002; Violin et al. 2011). Landscape-scale restoration is needed to create connectivity among populations. Methods used to promote connectivity may include increasing pond density across the landscape (Petranka and Holbrook 2006; Lesbarrères et al. 2010), and creating corridors in which dispersing amphibians and reptiles can bypass roads and other less-permeable land cover types (Aresco 2005; Woltz et al. 2008). The creation of large-scale vegetated corridors, such as urban greenways, may be particularly beneficial to aquatic and semiaquatic wildlife (Guzy et al. 2013, Chap. 12). Removal of human-created structures such as low-head dams and weirs can reestablish genetic exchange and allow anadromous and catadromous stream fishes to complete their life cycles (de Leaniz 2008). However, caution should be exercised as barriers to movement may be needed to prevent dispersal of invasive species (Thompson and Rahel 1998; Kerby et al. 2005).

16.5.3 Reintroduction and Translocation

Recovery of aquatic and semiaquatic wildlife populations in urban areas may involve reintroduction, repatriation, and translocation (RTT) of individuals. These methods are controversial management procedures and largely untested for aquatic and semiaquatic species (Dodd and Seigel 1991; Germano and Bishop 2008), thus criteria to evaluate the success rate (i.e., establishment of populations) of RTTs are lacking. Prior to RTT, several factors should be evaluated including hydroperiod, food availability, water quality, and the suite of competitors and predators in the receiving area (Semlitsch 2002). Aquatic habitats should have hydroperiods suitable for focal species or taxonomic group and lack introduced invertebrate and vertebrate predators. Food availability and water conditions (i.e., Sacerdote and King 2009) also need to be monitored prior to reintroduction to determine their suitability for a given species. Terrestrial upland habitat requirements should also be known for the reintroduced species. At minimum, wetlands should have surrounding buffers that include the critical upland habitat for reptiles and amphibians (Semlitsch and Bodie 2003), and appropriate BMPs to reduce flow variability, sedimentation, and chemical contamination should be in place prior to reintroduction. We advocate for the long-term monitoring of populations after RTTs to determine if populations become established.

16.5.4 Habitat Protection and Planning

In theory, prioritizing critical habitat and protecting habitat from degradation associated with urban development (e.g., invasive species, human subsidized predators, etc.) is the best way to manage semiaquatic and aquatic wildlife in urban and suburban areas (Chap. 12). For aquatic urban wildlife, critical habitat includes both aquatic and adjacent terrestrial environments (Semlitsch 2000; Semlitsch and Bodie 2003). Furthermore, to facilitate dispersal and continued functioning of population processes, connectivity among patches of critical habitat should be strongly considered (Semlitsch 2000). However, the land use within critical local habitats and landscapes varies in terms of permeability on a species by species basis. For example, semiaquatic turtles require open canopy uplands around aquatic environments to nest (Steen et al. 2006), whereas population persistence of some amphibian and fish species is determined by the extent of forested land surrounding wetlands, lakes, and streams (Homan et al. 2004; Francis and Schindler 2009). Thus, translating land preservation strategies to all aquatic and semiaquatic species is fraught with difficulty, as significant differences exist among fish, amphibian, and reptile species in terms of their habitat requirements.

Nonetheless, several general strategies in terms of land preservation and management can be applied to benefit the majority of species. First, land preservation strategies should be biologically based, such that the amount and type of land critical to the persistence of the local population is conserved. Findings by Semlitsch and Bodie (2003) indicated aquatic habitats should be buffered by 159 to 290 m of unfragmented, upland to protect wetland-breeding amphibians and 127 to 289 m to protect populations of semiaquatic reptiles. The effectiveness of critical habitat designations to protecting local populations of some semiaquatic species, especially in urban areas, may be questionable. Crawford and Semlitsch (2007) suggested 93 m of terrestrial buffer is required to protect stream-associated salamander populations; however, Willson and Dorcas (2003), Miller et al. (2007), and Roy et al. (2007) indicated that even small amounts of impervious surface cover (≥10%) within these stream catchments areas can have a profound negative impact on stream amphibian and fish populations.

Maintaining buffer zones around aquatic environments also serve to decrease the effects of urban and suburban areas on water quality and provides terrestrial subsidies to fish, amphibians, and reptiles inhabiting both lentic and lotic environments. If roads are near aquatic environments or located within critical habitat, proper measures, such as culverts or underpasses, should be incorporated and designed correctly to reduce mortality (Aresco 2005; Woltz et al. 2008, Chap. 15), and chemical treatments, particularly the use of road deicers, should be eliminated. Finally, critical features of habitat should not be removed or altered for aesthetic reasons. For example, deadwood and shoreline vegetation should be maintained in aquatic environments as these habitat features provide basking, breeding, and foraging sites for numerous species of aquatic and semiaquatic wildlife.

Implementing conservation through land purchase and protection in urban settings is often a costly endeavor, and thus comprehensive landscape planning that incorporates local knowledge of biodiversity "hotspots" is necessary to maximize the efficiency of funding. Thus, the first step in planning for land protection in urban and suburban settings should include a detailed inventory of habitats and species (see Chap. 12). Unfortunately, unless a species is protected by law (Buckley and Beebee 2004), knowledge of where these "hotspots" of aquatic and semiaquatic animal species exist is rarely available to or considered by landscape planners (Miller et al. 2009). When knowledge is lacking regarding sites of significant biodiversity, protecting habitats sensitive to urbanization, such as ephemeral wetlands and low-order streams should be priorities. These aquatic habitats have seen the sharpest level of decline and deterioration in urban and suburban settings (Rubbo and Kiesecker 2005; Roy et al. 2007; Elmore and Kaushal 2008), are known to be critical habitats for numerous fishes, amphibians, and reptiles, and will have positive impacts on regional hydrology and the water quality of downstream aquatic environments.

Local support for land conservation can be especially pervasive when coupled with recreation opportunities, such as those provided by greenways. In rapidly developing regions, the inclusion of green spaces has been common and they have been shown to counteract environmental impacts of urbanization (McPherson 1990; Rowntree and Nowak 1991; Simpson and McPherson 1996; Jim and Chen 2003), aid local economies by increasing property values (NPS (National Park Service) 2012), enhance the attractiveness of cities (Schroeder 1989), and play an important role in education (Rodenburg et al. 2002). Additionally, green spaces in urban areas can act as refuges for wildlife and aid in connectivity among populations (Terman

1997; Sodhi et al. 1999; Pirnat 2000; but see Garber and Burger 1995). However, knowledge of the effectiveness of green spaces in conserving populations of some taxonomic groups, such as semiaquatic animals, are generally lacking and/or restricted to certain types of green space, such as golf courses (See McDonough and Paton 2006; Harden et al. 2009; Foley et al. 2012; Puglis and Boone 2012; Guzy et al. 2013; Price et al. 2013).

Conclusion

Urban and suburban areas have a strong, usually negative, effects on aquatic environments, thus many species of semiaquatic and aquatic wildlife have experienced local extirpation or population declines in urban environments. However, a few species exhibit resistance to urbanization, and some may even thrive in urban and suburban aquatic environments. Regardless, in most regions, urban areas will continue to expand and degradation of aquatic environments will likely continue. General management strategies for semiaquatic and aquatic wildlife in urban areas require the use of proper stormwater treatment (including WTFs and BMPs), habitat restoration, potentially reintroductions or translocations, and sufficient planning to protect remaining critical habitats. These management strategies will not only protect aquatic and semiaquatic wildlife in urban and suburban areas, but also will benefit human inhabitants by conserving water quantity and quality.

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Chapter 17
Wildlife Damage Management in the Urban Landscape

David Drake

Keywords Humans · Wildlife · Interactions · Wildlife damage management

17.1 Introduction

Wildlife damage management is a growing field within the wildlife profession and uses science to guide management to reduce or eliminate problems created by wildlife. Wildlife damage occurs in all types of habitats and with a variety of species. However, because human and wildlife populations inhabit the same spaces in urban areas, and the urban landscape is novel compared to the environments in which animals evolved, managing wildlife damage in urban environments poses unique challenges.

17.2 Living Together: Humans and Wildlife

Wildlife damage management in urban areas is generally more complicated and difficult to manage relative to rural areas because of the human component. Humans and wildlife are concentrated in urban areas, so the possibility of human–wildlife interactions is greater, and more often than not, these interactions are perceived negatively. Urban residents, especially younger generations, spend more time indoors than out, and as a consequence, may not feel comfortable encountering wildlife (Fig. 17.1; Louv 2005). Thus, at least some of the interactions urban residents have with wildlife are perceived negatively due to unfamiliarity.

Attitudes about risks from wildlife vary considerably based on context and stakeholder group, and urban residents have varying degrees of acceptance or tolerance of different species of wildlife (Knuth et al. 2001). This tolerance is termed wildlife stakeholder acceptance capacity (WSAC) and is tempered by a stakeholder's

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D. Drake (✉)
Department of Forest and Wildlife Ecology, University of Wisconsin-Madison,
1630 Linden Dr., Madison, WI 53706, USA
e-mail: ddrake2@wisc.edu

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Fig. 17.1 The connection between humans, especially younger generations, and the outdoor world is diminishing. (Credit: USFWS/Steve Hillebrand)

perceived costs and benefits resulting from interactions with a particular species (Carpenter et al. 2000; Decker et al. 2001). For example, abundant white-tailed deer (*Odocoileus virginianus*) in suburban areas can result in deer–vehicle collisions and damage to landscape plantings, among other negative consequences. Many suburban stakeholders have a low tolerance for deer because they perceive greater risks or costs than benefits in their interactions with deer (Stout et al. 1993; Riley and Decker 2000).

In addition to negative perceptions about wildlife, many urban residents, somewhat unrealistically, expect wildlife related problems can be resolved immediately and at no cost. Few people are willing to accept waiting for wildlife damage problems to resolve themselves (e.g., noisy birds in a chimney leaving after their young are fledged). Similarly, few feel comfortable implementing the management options themselves, but most express frustration when they understand they may need to pay a professional to resolve the issue.

To further complicate wildlife damage management in urban areas, the traditional methods of regulating wildlife through hunting, and to a lesser degree trapping, are constrained in urban areas. As humans have increasingly become urban dwellers, dramatic decreases in the number of people hunting have occurred since the early 1980's (Robison and Ridenour 2012). This decrease in number of hunters in the USA not only results in fewer people available to harvest overabundant populations, but also means fewer people understand and support lethal solutions to urban wildlife problems. In addition to these societal shifts over time, people who oppose hunting tend to be concentrated in urban areas, making it more difficult, controversial, and expensive to manage wildlife damage in the urban context (Rutberg 2001; Conover 2002). Furthermore, as will be addressed later in this chapter, certain lethal methods are not always safe or legal in urban environments because of the high human population densities.

Fig. 17.2 Common wildlife species present in USA urban areas include: **a** white-tailed deer (credit: USFWS/Steve Hillebrand), **b** raccoon (credit: USFWS/Robert Burton), **c** Canada goose (credit: USFWS/Ryan Hagerty), and **d** hairy woodpecker (Credit: USFWS/Donna Dewhurst)

17.3 Characteristics of Problem Wildlife

Species that have caused the greatest economic damage to US metropolitan households are generalist species (i.e., urban exploiters and urban adapters) with behaviors that allow them to adapt to the unique food and cover resources in the urban environment. These species include mice and rats (*Rodentia*), squirrels (*Sciuridae*), raccoons (*Procyon lotor*), moles (*Talpidae*), European starlings (*Sturnus vulgaris*), and pigeons (*Columba livia*). Other species commonly involved in negative human–wildlife interactions in North American suburbs include American crows (*Corvus brachyrhynchos*), bats (*Chiroptera*), beaver (*Castor canadensis*), Canada geese (*Branta canadensis*), coyotes (*Canis latrans*), opossums (*Didelphis virginiana*), rabbits, (*Lagamorpha*), skunks (*Mephitidae*), wild turkey (*Meleagris gallopavo*), white-tailed deer, woodchucks (*Marmota monax*), and woodpeckers (*Picidae*) (Fig. 17.2;

Conover 1997, 2002). Nearly every species living in urban environments, however, has probably been considered a problem at some time.

The concentration of humans in urban areas have direct consequences on wildlife populations (Chap. 8), like those mentioned above, because urbanization creates highly fragmented landscapes as a result of infrastructure like roads and utilities, property boundaries, and mixed zoning. Fragmentation leads to an abundance of vegetation edges. Species commonly found in human-dominated environments tend to be edge oriented and habitat generalists, meaning they can exploit a variety of habitat types and food resources (Chap. 7). As such, they have benefited from the creation of abundant edge as urban environments are fragmented and developed to meet human needs (DeStefano and DeGraaf 2003).

The wildlife species most difficult to manage and control in urban environments are those most able to adapt to humans and human-dominated landscapes. For example, raccoons use cavities in trees for nesting and roosting, but will readily nest in chimneys because they are an artificial substitute for a natural cavity. Black bears (*Ursus americanus*) have been found hibernating for the winter under the porch or deck of suburban homes. Woodpeckers do not distinguish a tree from a wood-sided home as they search for insects and drum to defend territories. The more wildlife adapts to human-dominated habitats, the more they encounter, interact with, and habituate to humans. As habituation increases, wildlife lose their fear of humans and rather than flee when encountering a human, individuals or populations of wildlife will stay put (Chap. 9). In some instances, wildlife will become aggressive, especially during breeding activities when defense of territory, young, or mates are involved.

17.4 Human-Induced Habitat Element and Wildlife

Wildlife managers have struggled to control population growth for many common urban wildlife species. These species are particularly successful in urban areas due to favorable habitat (food, water, and cover) conditions (Chap. 8) created by human activities (Conover 1997; DeStefano and DeGraaf 2003). In addition to the naturally occurring habitat in urban areas, humans provide an abundance of supplemental food and cover in urbanized landscapes, both intentionally and unintentionally (Chap. 8). A good example of intentional provision of artificial food sources are bird feeders and landscape vegetation. But the unintended provision of habitat also commonly occurs. For example, barn swallows (*Hirundo rustica*) nest under the eaves of a house, and bats commonly roost or hibernate in attics.

In urban areas, human provided resources can allow wildlife populations to exceed biological carrying capacities reached (BCC) in natural areas. The BCC concept refers to the population size of a species that a particular environment can sustain. In urban areas, resources provided by humans (e.g., trash, supplemental food, landscape plantings) allow wildlife populations to be sustained beyond levels supported by natural food sources, and wildlife often over-eat the naturally available food sources in such contexts (McShea et al. 1997; Kilpatrick et al. 2001).

Cultural carrying capacity (CCC) may be more relevant in urban areas because it reflects the juxtaposition of wildlife and human populations in areas with dense concentrations of humans. CCC is the level of a particular wildlife population (e.g., deer, Canada geese) that is acceptable to the human population in a given area (Decker et al. 2001).

17.5 Economics of Damage and Health Risks

Wildlife damage can result in economic loss, health and disease issues, and ecological problems. Suburban and urban residents in the USA spend an estimated $ 8 billion annually due to wildlife damage in urban areas (Conover 1997). This figure accounts for money spent to reduce or eliminate wildlife damage and to fix or replace items damaged by wildlife, such as browse damage to landscape plantings and agricultural crops, and property damage from animals chewing and defecating on, and entering, residential and commercial buildings. The above figure does not take into account costs of wildlife–vehicle collisions that occur in urban areas. It is estimated that annual deer–vehicle collisions in the USA alone cause $ 4.6 billion in vehicle damage and health care costs, and many of these collisions occur in urban areas (Insurance Information Institute 2010). Moreover, the $ 4.6 billion figure does not consider collisions with other wildlife species.

Zoonotic diseases, or those that can be transmitted from animals to humans, can impact human, domestic animal, and wild animal health in the urban environment. Green spaces, such as greenways and other natural areas, are used by wildlife and by humans and their pets, and are potential hotbeds of zoonotic disease transmission (Dunbar et al. 2007). For example, rabies can be prevalent in some wildlife populations and is occasionally transmitted to humans and their pets (Blanton et al. 2006). Variants of the rabies disease occur in different parts of the USA, with bat (*Chiroptera* spp.), raccoon (*Procyon lotor*), skunk (*Mephitis mephitis*), and gray fox-variants (*Urocyon cinereoargenteus*) being the most widespread (Dunbar et al. 2007). These species serve as reservoirs for rabies, and commonly occur in urban areas.

Ecological problems caused by wild animals in the urban environment receive the least amount of attention because they do not necessarily impact humans in terms of economics or health, but can have substantial consequences in other ways. For example, it is estimated that there are as many as 100 million feral and abandoned cats (*Felis catus*) in the USA, many concentrated in urban areas (Lepczyk et al. 2003; Jessup 2004). Furthermore, there are millions more cats that are domestic and cared for by pet owners but are allowed outside (Lepczyk et al. 2003; Woods et al. 2003). It has been estimated that free-roaming and feral cats may kill 1.4–3.7 billion birds and 6.9–20.7 billion mammals in the USA annually (Loss et al. 2013). Lepczyk et al. (2003) found that at least two species of birds of conservation concern were depredated by cats, and suggested that cats can play an important role in fluctuating bird populations. Cat depredation is not limited to North America; it is a serious ecological problem impacting native wildlife throughout the globe (Barratt 1998; Woods et al. 2003).

17.6 Controlling Growth of Urban Wildlife Populations

Abundant habitat and adaptable wildlife species result in healthy, if not overabundant wildlife populations (Chap. 7). This becomes problematic given the limited ways to regulate population growth in the urban landscape. Typically hunting is not allowed in urban areas, and natural predators are often rare or replaced by feral and domestic cats and dogs (DeStefano and DeGraaf 2003). Furthermore, the public has demonstrated low tolerance for predators (e.g., wolves (*Canis lupus*), cougars) large enough to control wildlife like deer in the urban environment.

17.6.1 Lethal Management

Lethal management practices commonly used to kill individual animals include shooting, body-gripping (kill) traps, and toxicants. One of the primary advantages of lethal management is that it directly decreases the number of individuals that can potentially cause damage, thereby reducing the risk of damage for the short term. In cases where one or a few animals are causing the problem, lethal removal of the responsible individuals can completely eliminate the damage. A second advantage of lethal removal is that the suite of associated practices is more cost-efficient relative to many nonlethal practices (Conover 2002). Alternatively, the primary disadvantage of lethal practices is the lack of universal applicability. Because of high human densities in urban areas, much wildlife damage management is conducted in view of the public, and those that support lethal management may be in the minority. In fact, lethal management that is conducted in high visibility areas often courts controversy and protests (Knuth et al. 2001). Furthermore, in many urban communities across the USA, it is either not safe or illegal to kill wildlife using a firearm because too many humans live in close proximity.

Although a majority of the public does not trap nor understand the intricacies of trapping, it can be a cost-effective and successful tool to manage wildlife damage. Snap traps are a type of body-gripping trap commonly used to remove mice and rats in and around residential and commercial buildings. Other types of body-gripping traps are used for nocturnal and secretive furbearers (e.g., beavers, muskrats (*Ondatra zibethicus*)). For example, beavers create problems in urban areas by damming culvert pipes, thereby causing roadways to flood. Trapping is an efficient option to rid beavers from the area and eliminate the flooding. However, opposition to trapping increases dramatically when animals are viewed as charismatic or capture the public's interest. In fact, public engagement through ballot initiatives has outlawed some or all types of trapping in a number of US states (Decker et al. 2001).

The application of toxicants is the most common method for controlling rodents and other small mammals, but toxicants also are used on select bird species (Vercauteren et al. 2012). The advantage of toxicants is that they are cost-effective at

Fig. 17.3 Although it can be controversial in suburban settings, hunting is a cost-effective management option for certain species. (Credit: USFWS/Steve Hillebrand)

reducing large numbers of animals over a large area (Vercauteren et al. 2012). The disadvantage is that toxicants are not necessarily species-specific and can harm nontarget animals, so care must be taken when selecting the toxicant to be used and the application method (Vercauteren et al. 2012).

The use of traps and toxicants are effective for smaller animals and is often able to be done out of view of the public, but other lethal techniques, especially for larger animals, are not as discrete. For example, deer removal by shooting in suburban areas can be an expensive process compared to recreational (sport) hunting (Fig. 17.3). In many instances in suburban areas, controlled hunts and sharpshooting are used. Controlled hunting combines conventional deer hunting with more stringent controls and restrictions on hunter activities (e.g., controlled hunter densities, need to demonstrate proficiency). While controlled hunts commonly involve recreational hunters selected through a lottery system or some other selection process, sharpshooting involves either professional wildlife biologists or law enforcement officials. Both of these nontraditional hunting methods require cooperation from local law enforcement to enforce boundaries of the hunted area and prevent harassment of hunters. In addition, sharpshooting operations are granted exceptions not employed during controlled or recreational hunting opportunities, including hunting at night, from vehicles, shooting across roadways, and using night-vision scopes and noise suppressors (DeNicola et al. 1997).

In addition to white-tailed deer, Canada geese are commonly targeted for removal from urban areas of the USA. Nuisance Canada geese in the USA usually are rounded up in late spring and early summer when they are flightless and easy to catch due to molting of their flight feathers (Smith et al. 1999). Once rounded up, geese typically are placed in CO_2 chambers and euthanized. The meat from collected geese and deer is often donated to local food banks, but processing by volunteers or payment of processing fees may be required.

Fig. 17.4 Exclusion, often using fencing, is a popular nonlethal management technique to reduce or eliminate wildlife damage. (Credit: John Grande)

17.6.2 Nonlethal Management

Nonlethal management practices are intended to reduce or eliminate negative interactions with wildlife without killing the animals. This group of practices includes exclusion options (e.g., fencing), harassment (e.g., visual, noise, or both), repellents, trapping and relocation, habitat modification, and cultural modification (changing human behavior). The primary advantage of nonlethal practices is that they are more universally accepted and tolerated relative to lethal management. Nonlethal management practices, however, address wildlife damage by causing the animal(s) to move somewhere else. Hence, this approach may not address the problem but simply move it to a new location. For example, fencing off areas can concentrate animals on properties that have no fencing, thereby increasing wildlife damage for non-fenced landowners. Moreover, nonlethal practices are often more expensive to employ than are many of the lethal options, and are limited in their effectiveness because they do not remove individuals from a population, and therefore are not able to reduce the magnitude of the damage caused by that species (Conover 2002; Vercauteren et al. 2012).

Exclusion techniques are intended to prevent an animal(s) from accessing an area and are one of the few nonlethal solutions that provide long-term relief (Vercauteren et al. 2012). Exclusion methods include assorted forms of fencing, plastic netting, and overhead lines (Fig. 17.4). Exclusion practices do not always involve logistically difficult or expensive practices like fencing; they can include simple fixes like stuffing a hole in a building with steel wool or caulking to prevent access to mice, bats, and squirrels.

Harassment techniques are intended to cause animals to not feel safe in an area where these techniques are being employed, and therefore, abandon, or lessen their use, of the area. Harassment techniques, also referred to as frightening or hazing agents, include visual options like flashing lights, lasers, mylar balloons, kites and flagging, and predator effigies (e.g., fake owl or coyote), noise options like propane cannons, shell crackers, or other noise making devices or a combination of visual and noise harassment like dogs and inflatable human effigies with integrated noises like sirens. Harassment techniques that combine visual and noise options often are more effective than if a visual or noise technique is used alone (Booth 1994). However, animals can quickly habituate to harassment devices, and once that happens, the harassment technique is no longer effective. In addition, noise devices may have limited applicability in urban settings because of the disturbance they cause to humans.

Repellents are chemicals or substances that cause an adverse reaction for any animal that comes into contact with them. The strength of the negative experience can determine the effectiveness of the repellent. The two most common types of repellents are contact (aka taste) or area (aka odor) repellents. Contact repellents are a liquid formulation with different active ingredients depending on the brand of repellent, and are sprayed on vegetation that requires protection. Contact repellents are effective through taste aversion. Area repellents repel animals from an area by mimicking predator odors (e.g., coyote urine) or a repugnant smell (e.g., rotten eggs). Contact and area repellents are most effective on mammals because they have a more well-developed sense of taste and smell relative to other taxa of wildlife (Evans and Heiser 2004). The effectiveness of contact and area repellents varies based on population density of the targeted wildlife species, the weather, and time of year of application, among other factors (Conover 2002).

Trapping and relocating problem wildlife can be effective on an individual animal basis, but is problematic when trapping large numbers of animals within a species. Trapping and relocating individual animals involves capturing them in a live trap and then relocating them to a suitable release site away from the problem area, and far enough away that they do not return to the area from where they were trapped (Conover 2002). However, it is difficult to find areas willing to accept large numbers of potentially problematic animals for translocation, especially in urban areas, and in some cases, it is illegal to move animals away from trapping locations because of concerns about disease spread (e.g., deer and chronic wasting disease). In addition, a relocated problem animal can potentially cause problems at a different location. Another consideration is the welfare of the trapped and released animal. Live trapping can cause death via shock, and releasing an animal (especially one that is territorial) into unfamiliar territory can increase stress and the probability of death (Conover 2002).

Habitat modification involves changing habitat features used by the species causing problems in a way that makes the habitat less attractive to discourage the animal(s) from spending time in the area. However, most of the species causing damage in urban areas are habitat generalists, so altering habitat conditions is less effective at deterring urban wildlife relative to other options. Furthermore, wildlife habitat in human-dominated landscapes is already highly fragmented and modified, and wildlife damage problems continue to occur. More importantly, perhaps, habitat modification intended to discourage one problem species can create another problem species unless implications for multiple species are considered simultaneously.

Cultural modification is the final nonlethal management practice, and compared to the other choices, may be the most difficult to implement because this option involves changing the behavior of humans to prevent or mitigate negative human–wildlife interactions. For example, many things we do as humans attract animals. The most common attractant is food, be it feeding a domestic pet outdoors where wildlife occur, placing trash outside before it is to be collected, or storing birdseed where it is available to rodents. This type of cultural modification prevents problems rather than addressing them. However, just as wildlife are creatures of habit, so too are humans, and it is difficult to change human behavior even if it would result in eliminating or reducing wildlife damage.

17.6.3 Nontraditional Management

Because of the challenges of managing wildlife damage in the urban environment, some non-traditional options have been explored or proposed. Two examples are fertility control and commercial harvest.

17.6.3.1 Fertility Control

The objective of fertility control is to reduce the fertility of a wildlife population to reduce or eliminate growth of the population, and therefore, reduce the magnitude of the damage caused by the population. There are three methods for controlling reproduction, primarily aimed at mammals—contraception, sterilization, and contragestation. For controlling avian reproduction (primarily aimed at Canada geese in the USA), egg addling, oiling, or puncturing are widely used, and a product branded Ovocontrol® is available to control egg hatchability.

Contraceptives include chemosterilants and intrauterine devices, but the most commonly employed contraceptives in the urban landscape are immunocontraceptives. Immunocontraceptives control fertility by stimulating the production of antibodies that interfere with proteins and hormones necessary for reproduction (Miller et al. 1998). Immunocontraceptives are remotely delivered, either through a hollow dart or biobullet shot from a gun and containing an antifertility drug, or through treated bait piles. Of the three reproductive control options currently available, sterilization is the only one that provides permanent fertility control. In females, sterilization entails tubal ligation or the removal of the ovaries, and in males, either a castration or vasectomy is performed. Contragestation is the process of administering a drug after contraception to terminate pregnancy, and is used very infrequently (Miller et al. 1998). Contragestation drugs are administered remotely, similar to immunocontraceptive agents.

A number of techniques are available to prevent bird eggs from hatching, and thus reducing the number of individuals recruited into the population. These techniques are used in urban areas primarily to reduce overabundant Canada goose populations. Canada geese are indeterminate layers, meaning if a goose nest or the eggs within are destroyed, especially early in the incubation period, the female goose will establish a second nest and lay another clutch of eggs. However, the embryo inside an egg may be destroyed while leaving the eggshell intact, thereby “fooling” the female goose into continuing to incubate the intact eggs. When she finally abandons the nest because no goslings hatched out, it is late in the nesting season and she does not establish a second nest. Embryos may be destroyed by addling (shaking the egg vigorously), coating the egg with heavy mineral oil, or puncturing the egg. Coating the egg with heavy mineral oil prevents oxygen and moisture exchange between the inside and outside of the egg, thereby stopping embryo development. Puncturing eggs is done with a nail or other sharp instrument and is effective because it prevents a proper incubation temperature from being achieved, stopping embryo development. In addition to methods to destroy the embryo, Ovocontrol® is a United States Environmental Protection Agency-registered product available to prevent yolk development in a number of avian species, including Canada geese. Ovocontrol is administered through a treated bait pile.

The primary advantage of reproductive controls is that they provide a nonlethal option for population reduction. However, the disadvantages may outweigh this advantage. While demonstrated to be effective on individual animals, cost-effective population reduction via reproductive controls has not been demonstrated in most cases on a broader, landscape and population level. In addition, because many of the reproductive control methods require retreatment and/or vigilance, they are logistically difficult and expensive to employ.

17.6.3.2 Commercial Harvest

A new conceptual idea at this time, with the potential for reducing overabundant game species like deer, is a regulated commercial harvest. Vercauteren et al. (2011) proposed allowing hunters who have demonstrated safety and proficiency to harvest deer in a defined and monitored area, and then sell the harvested animals on the market for a profit. A regulated commercial harvest may be most feasible in suburban areas that have experienced long-term deer damage (e.g., browse of landscape plantings, deer–vehicle collisions) and that are not able to reduce the deer population to an acceptable level with currently available tools. In many suburban areas, the lack of hunters prevents deer (and potentially other game) populations from being reduced in a cost-efficient manner. The ability to sell harvested deer may provide an economic incentive to entice hunters to apply greater harvest pressure at little cost to the local community. Currently, a regulated commercial harvest is not legal within the USA. However, if state laws are amended, such commercial hunts could be used to reduce deer populations and associated negative impacts, and provide a source of natural and healthy, locally produced protein (Vercauteren et al. 2011).

Conclusion

The human population is projected to continue its growth at least throughout the twenty-first century. As this occurs, more land will be urbanized to meet the needs of the human population, and it is anticipated that human–wildlife interactions will increase. It is important that the human population understands the difference between perceived and real human–wildlife problems and increases their tolerance for wildlife inhabiting the same landscape as humans. At the same time, education, management, and research efforts must continue to improve the efficacy and cost-effectiveness of current management options while continually seeking new solutions.

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