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Chapter 16
Managing Aquatic Environments for Wildlife in Urban Areas

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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.

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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 ($\geq 10\,\%$) 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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