Urban Development and Environmental Degradation

Wayne C. Zipperer, Robert Northrop, and Michael Andreu

Subject: Ecology, Sustainability and Solutions Online Publication Date: Aug 2020
DOI: 10.1093/acrefore/9780199389414.013.97

Summary and Keywords

At the beginning of the 21st century more than 50% of the world's population lived in cities. By 2050, this percentage will exceed 60%, with the majority of growth occurring in Asia and Africa. As of 2020 there are 31 megacities, cities whose population exceeds 10 million, and 987 smaller cities whose populations are greater than 500 thousand but less than 5 million in the world. By 2030 there will be more than 41 megacities and 1290 smaller cities. However, not all cities are growing. In fact, shrinking cities, those whose populations are declining, occur throughout the world. Factors contributing to population decline include changes in the economy, low fertility rates, and catastrophic events. Population growth places extraordinary demand for natural resources and exceptional stress on natural systems. For example, over 13 million hectares of forest land are converted to agriculture, urban land use, and industrial forestry annually. This deforestation significantly affects both hydrologic systems and territorial habitats. Hydrologically, urbanization creates a condition called urban stream syndrome. The increase in storm runoff, caused by urbanization through the addition of impervious surfaces, alters stream flow, morphology, temperature, and water quantity and quality. In addition, leaky sewer lines and septic systems as well as the lack of sanitation systems contribute significant amounts of nutrients and organic contaminants such as pharmaceuticals, caffeine, and detergents. Ecologically, these stressors and contaminants significantly affect aquatic flora and fauna.

Habitat loss is the greatest threat to biodiversity. Urbanization not only destroys and fragments habitats but also alters the environment itself. For example, deforestation and fragmentation of forest lands lead to the degradation and loss of forest interior habitat as well as creating forest edge habitat. These changes shift species composition and abundance from urban avoiders to urban dwellers. In addition, roads and other urban features isolate populations causing local extinctions, limit dispersal among populations, increase mortality rates, and aid in the movement of invasive species. Cities often have higher ambient temperatures than rural areas, a phenomenon called the urban heat island effect. The urban heat island effect alters precipitation patterns, increases ozone production (especially during the summer), modifies biogeochemical processes, and causes stresses on humans and native species.

The negative effect of the expansion and urbanization itself can be minimized through proper planning and design. Planning with nature is not new but it has only recently been recognized that human survival is predicated on coexisting with biodiversity and native communities. How and if cities apply recommendations for sustainability depends entirely on the people themselves.

Keywords: urbanization, megacities, hydrology, deforestation, urban heat island, planning

Cities: Present and Future

When compared with other landscapes such as forest, grasslands, wetlands, and deserts, the urban landscape is unique in that it is totally novel. The urban landscape evolved only recently in geological time: dates range from 6000–11,000 years ago depending on one's definition of a city. Since their inception, cities have changed from agricultural and trading hubs to large sprawling metropolitan regions. While modern cities are the economic engines and cultural centers of the world, they have significant effects on the environment and the ecological services that humans derive from natural systems.

This article will examine the projected growth of the urban population, infrastructural needs for that growth, and the potential ecological consequences caused by that growth if infrastructure is not planned appropriately. Specifically, the article focuses how projected urbanization will affect hydrologic and terrestrial systems with an emphasis on hydrology and biodiversity.

But, first, what is a city? How does one define a city? Currently, there is no standard global definition of a city. Instead there are three concepts—city proper, city agglomeration, and metropolitan area. City proper generally refers to the administrative boundaries of a city. Agglomeration refers to the area of contiguous urban cover, which is also known as the built-up area. By comparison, a metropolitan area encompasses both the city proper and agglomeration as well as adjacent areas that are economically and socially connected to the administrative city. Metropolitan areas can also contain non-urban lands (e.g., farms and forests). Because city proper does not capture all the built-up area and metropolitan areas contain non-urban land use, the city agglomeration is often used to define a city globally (United Nations, 2016).

In 1900, the world's population was 1.6 billion people, with approximately 13% living in cities. By 2008, urban populations had grown to 50% of the world's population, with more than 3.9 billion people living in urban areas. By 2050, urban populations are expected to increase by 2.5 billion and exceed 60% of the world's population (United Nations, 2014; Figure 1). Most of this population growth will occur in Asia and Africa, with 37% of that growth occurring in just three countries: China, India, and Nigeria.

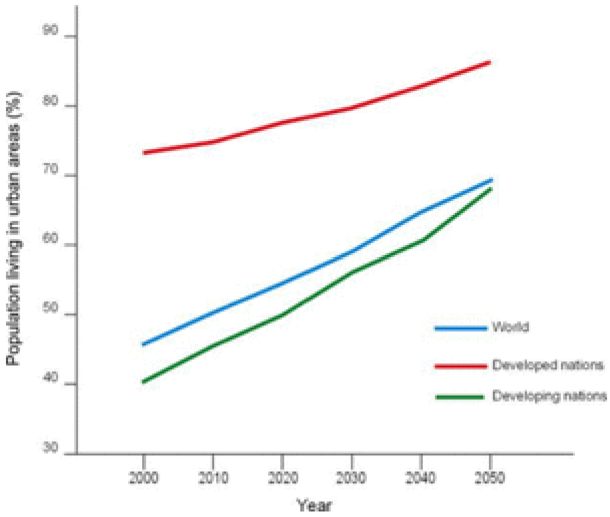

Figure 1. Projected urban population growth to 2050 for the world, developed and developing nations.

Cities with populations over 10 million are called megacities. In 1950, there were only two in the world, Tokyo and New York City. Today, there are 31 megacities with the majority of them occurring in Asia. Tokyo, Japan, is the largest city in the world when considering size and population. The agglomerated city spans 13,500 km^2 (5200 mi^2) and has a population of more than 38 million individuals. By 2030, more than 41 megacities are projected to exist. They will be home to more than 730 million individuals (United Nations, 2016).

Urban population growth is not restricted just to megacities. In fact, the majority of urban population growth will occur in cities of fewer than 5 million. Currently, there are 436 cities with populations between 1 and 5 million and 551 cities with populations between 500,000 and 1 million inhabitants. By 2030, those numbers are expected to increase to 559 and 731, respectively (United Nations, 2016).

With that said, not all cities are expanding. Some are actually becoming smaller and are referred to as shrinking cities. Because of changes in the economy, low fertility rates, lack of immigration, and catastrophic events such as a hurricane, a city could lose population. The loss of population creates new social as well as ecological challenges. Economics, either from resource depletion or changes in technology, play an important role in population growth or loss in a city. For instances, during the 1950s and 1960s, the city of Detroit, Michigan grew rapidly because of the auto industry. Changes in that industry and global markets during the 1970s as well as segregation and the white population moving to suburbs resulted in a 60% decline in the population of Detroit, from 1.8 million in 1950 to 713,777 in 2010. Similarly, Daegu, South Korea, was a booming city as the textile hub of the country during the 1960s. With globalization, it suffered economically, which precipitated a population decline and the decay of its city center (Joo & Seo, 2018). Other cities in Asia have also experienced growth and decline associated with economic boom and bust cycles of resource extraction (Biswas, Tortajada, & Stavenjagen, 2018).

For a city's population to grow, a birth rate of greater than 2.1 individuals per female is needed. When rates are less than that, the population is not replacing itself. Cities in Europe, the United States, Australia, and Asia are losing populations because of low fertility rates. For instance, in South Korea and Japan fertility rates are 1.17 and 1.4 individuals per female, respectively (Biswas et al., 2018). In Germany, fertility rates have declined from more than 2.5 individuals per female during the 1960s to 1.57 individuals per female in 2017 (Knoema, 2020).

Birth rates are not the only way for a population to increase. Immigration into a city can increase population by adding more people and contributing to the overall birth rate. In fact, immigration can play an important role in the growth of a city and a nation. The influx of immigrants, however, can create its own set of social challenges with integration and assimilation, as witnessed in Europe in 2015 (Milanski, 2019).

Managing infrastructure in shrinking cities is a major planning problem. Unused buildings are often abandoned and left to decay, becoming a visual blight. Demolishing abandoned buildings could create publicly owned vacant lots, which, when managed properly, could provide ecological benefits in the form of gardens and parks. Unfortunately, shrinking cities often lack the financial resources to develop and manage vacant lots in an ecologically or socially sound manner. Driven by blighted conditions and lack of economic opportunities, a city can spiral into a positive feedback loop of continued population loss, which causes further decline economically, socially, and ecologically. Often multiple policies and actions are needed to reverse the course of a city in decline to a city inrecovery (Mallach, Haase, & Hattori, 2017).

Catastrophic events, such as a hurricane or tsunami, can cause a significant decline in population. For instance, prior to Hurricane Katrina (2005), New Orleans, Louisiana, had a population of 484,674. In 2018 its population was 386,617, a 20% decline. Not only did the storm displace people, but it also changed the ecology of the city. Residential lots once occupied by houses and maintained by individuals before the storm, are now occupied by unmanaged vegetation, often dominated by invasive species (Lewis et al., 2017). A similar scenario may be playing out in Puerto Rico due to Hurricane Maria.

The infrastructure required to manage future population growth in most urban landscapes is huge. Seto, Güneralp, and Hutra (2012) has estimated that global urban expansion is two times what is actually needed to house future population growth. The additional expansion results from commercial, institutional, transportation, and industrial needs. Current projections require a global expenditure of $5 trillion per year to meet infrastructural needs in water, agriculture, telecommunication, power, transportation, buildings, and industries (Forum, 2013). In the United States, Nelson (2004) reports that 33% of residential structures needed to house projected population growth in the United States do not currently exist. The environmental degradation can be considerable if this development is not planned sustainably, which involves the collective integration of ecological, social, and economic components of the city.

Urbanization's Effect on Hydrologic Systems

If not planned properly, the projected infrastructural needs to support anticipated population growth will have a devastating effect on the hydrologic processes of a city and surrounding region. Urbanization alters hydrologic processes in a number of ways including reducing infiltration, increasing storm water runoff, altering hydrologic pathways and stream flows, modifying stream geomorphology and stream ecology, and decreasing water quality and quantity. This section will examine how urbanization affects the hydrologic system.

Before examining how urbanization affects hydrologic systems, a basic understanding of hydrology is needed. The watershed, an area drained by a stream, is the fundamental unit of measurement (Sun & Lockaby, 2012). There are four key processes of a watershed that affect the hydrologic cycle: precipitation (P), evapotranspiration (ET), stream flow (Q), and water storage (S). These elements interact in the following way:

$$\Delta S = P - \mathrm{ET} - Q$$

(1)

where ΔS is the change in water storage, the amount of water stored in the watershed. Precipitation has the largest input. Evapotranspiration results from the evaporation of water from soil surfaces and transpiration from vegetation. In general, higher vegetation cover results in higher rates of ET. Stream flow is the amount of water flowing out of the watershed and is a mix of three flow components—overland, subsurface, and groundwater. Overland flow, also known as runoff, is the amount of water that flows on the land surface and is related to the intensity of rainfall and soil conditions. When an intense rain event occurs and soils are saturated or unable to absorb the rainfall, the water will move across the land surface to the stream. By comparison, subsurface flow is the water that flows beneath the surface into the stream. Groundwater flow is the flow of underground water that is held in the soil or in rock crevices (Sun & Lockaby, 2012).

Using a forested watershed (one could also use grass or brush lands) as a point of reference, water enters the watershed principally through precipitation. Because the watershed is forested, there is a high loss of water through transpiration, minimal overland flow, and higher subsurface and groundwater flows. The low overland flow results from a high infiltration rate of the precipitation reaching the ground. In general, 40% of the precipitation is lost through evapotranspiration, 10% through runoff, 25% through shallow infiltration (which contributes to subsurface flows), and 25% to deep infiltration (which contributes to groundwater flow) (Arnold & Gibbons, 1996; Figure 2). Actual values will vary across different regions because of differences in soil texture, temperature regimes, and precipitation patterns.

To illustrate the effect of urbanization on hydrologic systems, the hydrology elements P, ET, Q, and ΔS will be used. A watershed with vegetation cover will be used as a starting point, and it will be assumed that precipitation patterns do not change with urbanization.

Typically the first effect of urbanization is the removal of vegetation. As one may expect, ET declines because of the loss of transpiration from the vegetation, but the loss is less than expected (Figure 2). Urbanization adds impervious surfaces through the construction of buildings and roads. Water evaporates from these surfaces too. Assuming a 10–20% cover of impervious surface, ET would decline only to 38%. Both subsurface and groundwater flows also decline because of less infiltration due to impervious surfaces. On the other hand, runoff increases to 20% (Arnold & Gibbons, 1996). Increasing impervious surface cover to 30–50% has the following effect. Vegetation cover declines as more land is covered by an impervious surface, which results in ET declining to 35%, shallow infiltration decreasing to 20%, and deep infiltration decreasing to 15%. Runoff, on the other hand, increases to 30%. In other words, 20% of the precipitation entering the watershed is moving directly into the stream as overland flow. As impervious surfaces increases to 75–100%, ET declines to only 30% (again, water evaporating from impervious surfaces contributes ET even though vegetation cover is minimal). Both shallow and deep infiltration rates are sharply decreased (10 and 5%, respectively), whereas runoff now increases to 55% (Arnold & Gibbons, 1996).

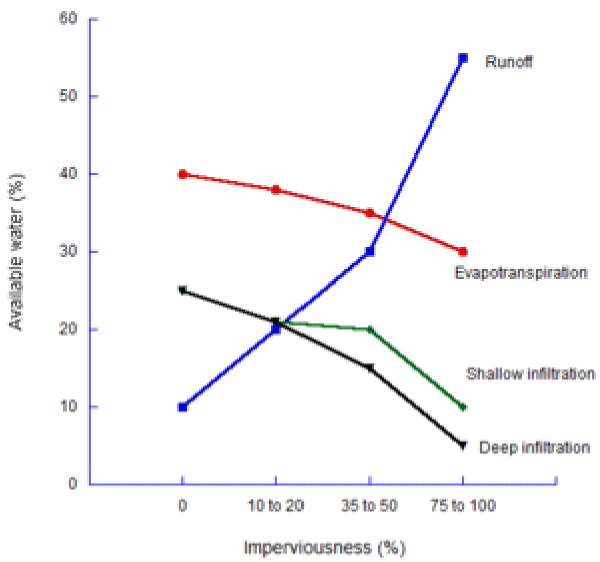

Figure 2. Graphical representation of changes in runoff, evapotranspiration, shallow and deep infiltration as percent impervious increases (adapted from Arnold & Gibbons, 1996).

Urban stream syndrome is a term used to describe the effect of urbanization on streams (Table 1). The increase in runoff or overland flow significantly alters stream flow, geomorphology, and ecology and water quality and quantity (Paul & Meyer, 2001). With just a 10% increase in impervious surfaces in a watershed, runoff increases, which alters peak flow, with the high flow rate of the stream as illustrated in a stream hydrograph (Figure 3). A stream hydrograph shows how stream flow changes over time with respect to a rain event. In a forested condition, peak flows are moderated by high infiltration rates, thus the frequency of flooding is attenuated. In comparison, with an increase in impervious surfaces, there is less infiltration and more water entering directly into the stream, causing higher peak flows to occur shortly after a rain event (Figure 3). The hydrograph also illustrates that base flow is lower in urban than in forested watersheds. The reduction of base flow lowers the depth of groundwater, a condition called hydrologic drought (Groffman et al., 2003).

Table 1. Generalized Effects of Urbanizing Land Use on Urban Streams (adapted from Walsh et al., 2005). Effects can either be an increase (↑) or a decrease (↓).

Hydrology	↑Frequency of overland flow
	↑Frequency of erosive flow
	↑Magnitude of high flow
	↓Lag time to peak flow
	↑Rise and fall of storm hydrograph
Water chemistry	↑Nutrients (N, P)
	↑Toxicants
	↑Temperature
Channel morphology	↑Channel width
	↑Pool depth
	↑Scour
	↓Channel complexity
Fishes	↓Sensitive fish
	↑Tolerant fish
Macroinvertebrates	↓Sensitive invertebrates
	↑Tolerant invertebrates
Algae	↑Eutrophic diatoms
	↓Oligotrophic diatoms
Ecosystem processes	↓Nutrient uptake

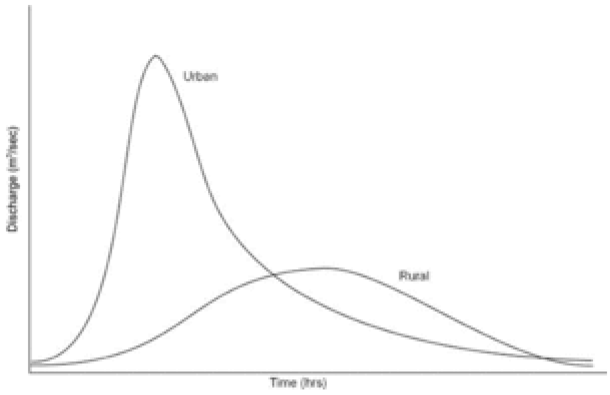

Figure 3. A hypothetical hydrograph of a forested landscape and an urban landscape showing the differences in peak flow rates because of differences in infiltration.

Higher peak flows can also change stream bank morphology and the ecological integrity of the aquatic system. As peak rates increase, the velocity of flow also increases. This additional energy in stream flow can contribute to stream bank erosion and incise the stream, thus making the stream bed wider and deeper. The erosion adds sediments to the stream, which can destroy habitat for aquatic fauna such as macroinvertebrates and fishes. The reduced base flow also significantly affects aquatic fauna and their habitats. Consequently, the number of species and abundance of sensitive species often decline as the land use surrounding streams becomes more urbanized. In comparison, some aquatic species that are tolerant to changes caused by urbanization may actually increase in abundance.

With the addition of impervious surfaces in the watershed, the temperature of the water in streams can increase with runoff. Impervious surfaces can store solar energy as latent heat. As water flows over these surfaces, it heats up. When this warmer runoff enters the stream it can elevate stream temperatures. During a summer storm, runoff can cause stream temperatures to increase as much as 10–15°C (Paul & Meyer, 2001). In addition, riparian vegetation is often removed during the conversion of a forested site to urban land use. The loss of forest cover reduces the amount of stream shading and exposes the stream to more solar heating during the day and rapid cooling at night. These altered fluctuations in temperature can have a significant effect on biogeochemical reactions such as litter decomposition as well as the metabolism of aquatic organisms.

As runoff moves across urban surfaces it can pick up contaminants such as heavy metals, biocides (i.e., fungicides, pesticides, and insecticides), nutrients, and ions. These contaminants affect water quality. For instance, in temperate climates, salt is used as a de-icer during winter months. The sodium chloride ions ultimately end up in streams, changing the water chemistry and potentially affecting aquatic organisms. Leaky sewers and poorly maintained septic systems as well as the lack of sanitation systems can contribute significant amounts of nutrients into streams. In addition, organic contaminants such as pharmaceuticals (e.g., personal care products, antibiotics, reproductive hormones from birth control substances) and other organic compounds such as caffeine and detergents enter the stream because of faulty sanitation systems (Kolpin et al., 2002). Nutrient (N, P, K) loading also occurs through lawn fertilization. Fertilizers can flow off the land with runoff or leach into groundwater where they can be carried into water bodies. Similarly, heavy metals from brake linings and tires accumulate on roads and parking lots. Runoff from these surfaces often carry zinc, nickel, copper, lead, and chrome into nearby streams, where metal ions bond with organic material and settle into the sediments (Paul & Meyer, 2001). These sediments can be toxic and persist for long periods of time in the aquatic ecosystem.

The detection of organic contaminants in streams and rivers is a relatively recent phenomenon (Kolpin et al., 2002) and depends on a city's or nation's drinking water standards. Nonetheless, these contaminants pose a significant threat to aquatic flora and fauna. For instance, exposed microbial communities are being altered or developing resistance to organic contaminants, especially antibiotics (Rosi et al., 2018). Fish communities show signs of altered sex ratios and/or sexual development. Unfortunately, current wastewater treatment facilities and septic systems do not treat or remove these substances. Consequently, the long-term effect on aquatic flora and fauna is relatively unknown.

Cumulatively, these urbanization effects on stream geomorphology and water quality and quantity have tremendous effects on the hydrologic system and the viability of aquatic plants and animals (Table 1). Technology exists to modify or minimize the urban stream syndrome and restore ecological structure and function. One of the first steps towards improving ecological structure and function is to decrease peak flow and increase baseflow (Bernhadrdt & Palmer, 2007). This can be achieved by improving infiltration through storm water management. Storm water runoff can be retained, detained, slowed, or redirected to reduce peak flows through the use of storm water basins and constructed wetlands. Both of these practices can be designed also to enhance biological diversity. Once peak flow has been mitigated, stream channel stability can be restored through stream bank restoration and by replanting riparian vegetation. With the improvement in stream morphology and the addition of riparian vegetation, habitats for aquatic organisms are also improved.

Urbanization's Effect on Terrestrial Systems

Urbanization affects terrestrial systems both directly and indirectly. Examples of direct effects include land use conversions and habitat fragmentation. Examples of indirect effects include the urban heat island, introduction of invasive species, and altered biogeochemical processes. Cumulatively, these effects significantly decrease native species diversity as urbanization intensifies (Sukopp, 2004). This section, examines each of these effects on biodiversity.

Habitat Fragmentation and Conversion

The greatest threat to biodiversity is habitat loss. As of 2020 approximately 13 million hectares of forest land are lost annually to agriculture, urbanization, and industrial forestry (DeFries, Rudel, Uriarte, & Hansen, 2010). The loss of forested lands is primarily due to increases in human population, the need for greater agricultural production, and newly built infrastructure. With continued urban population growth, continued habitat alteration, degradation, and loss can be expected. These effects will have a significant impact on biodiversity.

. Direct loss has multiple effects. First, it decreases the amount of habitat, which can significantly affect those species (e.g., large predators) requiring large areas of habitat. Second, it increases habitat isolation. Habitats become embedded in a sea of agriculture or urbanization. Further isolation by roads and fences makes it difficult, if not impossible, for animal species to move from one habitat to another. Third, the habitat itself changes because of changes in the environment and habitat modification. The result is a degradation of habitat quality and subsequent changes in biodiversity. The habitat no longer supports native species richness (number of species) and composition (types of species) before landscape modifications.

Because of the significant amount of literature on deforestation and forest fragmentation, forest lands will be used as an example of how urbanization alters a habitat. The examples given are also applicable to grasslands and shrub lands. With deforestation and forest fragmentation, forest interiors are lost and forest edges are created (Figure 4). Forest patches that are greater than 5 hectares (ha) in size can contain forest interior (Levenson, 1981). Generally, forest interiors are mesic (moist) environments dominated by species that are shade-tolerant (flora) and with specific niche requirements (fauna). With deforestation and fragmentation, the size of a forest patch decreases, which reduces the amount of forest interior, but also alters the amount of edge habitat. Below 5 ha in size, a forest patch is basically edge habitat (Levenson, 1981). Edge habitats are more xeric than interior habitats because of exposure to solar radiation and wind. Consequently, with development of forest edges, species composition shifts to shade-intolerant species (flora) and species that are generalists (fauna)—adapted to a number of different types of habitats.

Figure 4. Deforestation and fragmentation patterns of forest patches and corresponding areas of interior and edge habitat (adapted from Zipperer, 1993).

The effect of deforestation and fragmentation on forest interior and edge depends on the clearing size, shape, and location (Zipperer, 1993) (Figure 4). For example, in a large block of forest habitat, an internal pattern of deforestation would destroy forest interior habitat while increasing forest edge habitat. By comparison, an external pattern of deforestation can cut into the block of forest land, thereby decreasing its area while at the same time increasing or decreasing forest edge. Depending on the pattern of deforestation, the amount of forest edge may increase or decrease. Regardless of the deforestation pattern, habitat suitability for native species requiring interior habitat is degraded. There is less habitat for interior species, and with the increase in forest edges there is a corresponding increase in the probability of invasive species spreading into the forest itself.

The effect of deforestation and fragmentation from urbanization has been extensively studied for birds. These studies have shown that species richness and abundance of forest interior species, primarily neo-tropical migrants, decline as a result of an increase in deforestation and fragmentation. Deforestation and fragmentation also increase the rate of brood parasitism by Brown-headed cowbirds (*Molothrus ater*) and rate of nest predation, both associated with increased edge habitat (Robinson, Thompson, Donovan, Whitehead, & Faaborg, 1995). Forest interior species are called urban avoiders, species generally absent from the urban landscape, but can be present in large natural areas embedded in cities. By comparison, species abundance of edge-tolerant species, such as cardinals (*Cardinalis cardinalis*), American crows (*Corvus brachyrhynchos*), and blue jays (*Cynacitta cristata*), increases with edge habitat and urbanization. These species are urban dwellers, species that can reproduce and persist in urban areas (Fischer, Schneider, Ahlers, & Miller, 2015). In fact, the suburban and urban landscapes have created new habitats, such as lawns, hedgerows, and buildings. These novel habitats can increase the abundance of a number of species such as house sparrows (*Passer domesticus*), European starlings (*Sternus vulgaris*), pigeons (*Columba livia*), and some species of swallows.

Urban structures themselves can have both significant positive and negative effects on birds. For instance, peregrine falcon populations have increased because the species uses high rises in cities as nest sites and perches. The falcons feed on the pigeons in the city. In contrast, buildings have caused a significant number of bird deaths. Between 365 and 988 million birds are killed each year in the United States because of collisions with building (Loss, Will, Loss, & Marra, 2014). Interestingly, the majority of deaths result from residential (45%, one to three stories high) and low-rise (55%, four to 11 stories high), and not high-rise structures (>11 stories high).

Roads, built to support urbanization, fragment the landscape, which has multiple effects on terrestrial biodiversity (Forman & Alexander, 1998). First, roads fragment existing habitats by literally dividing a habitat into two sections. Second, roads isolate species populations and impede migrations and dispersal. For example, amphibians and reptiles often have to cross roads to migrate to breeding habitats, resulting in significant mortality during the breeding season. Similarly, fragmented populations may experience limited genetic flow because of road avoidance. Some mammal species just do not approach or cross roads, thus limiting opportunities for genetic exchange and the potential for inbreeding. Even plants can be affected. The dispersal of some fleshy fruits and sticky seeds that are dispersed by mammals can be limited if their dispersal agents will not cross a road (Santos & Telleria, 1994). Natural local extinctions within the isolated fragments lead to species impoverishment, as species are no longer able to bridge road barriers and recolonize sites they once occupied. Third, roads create pathways for the movement of non-native, invasive species, with vehicles serving as dispersal agents (von der Lippe & Kowarik, 2008). Finally, roads alter environmental flows such as hydrological pathways and increase pollutants such as heavy metals, oils, and other fluids as well as the flow of salts and sediments into adjacent streams. As a consequence, only those species that are able to move through the road barriers and tolerate the fragmented habitat will increase in abundance and distribution.

Urban Heat Island Effect

The loss of vegetation cover and the addition of built structures affects not only habitat availability and species composition but also the very environment itself. The replacement of vegetation cover with impervious surfaces such as roads, parking lots, and buildings increases the amount of heat-absorbing surfaces. The increase in surface temperatures and the slow release of that heat ultimately increases ambient temperatures. Furthermore, the heat generated from fuel composition and air conditioning adds heat to the environment. Consequently, air temperatures in cities can be as much as 5°C warmer during the day and as much as 11°C warmer at night than rural areas. This phenomenon is known as the urban heat island effect (see EPA, 2008; Figure 5).

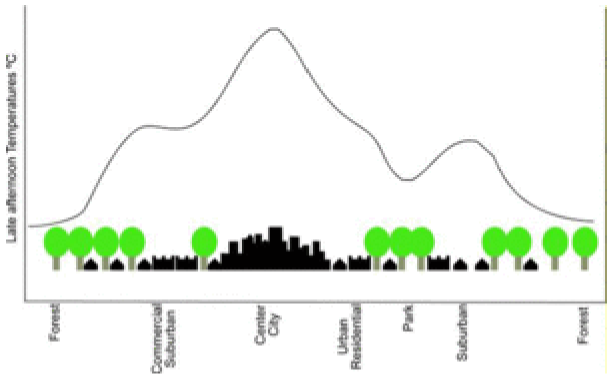

Figure 5. A possible temperature profile from center city to the rural landscape illustrating the urban heat island effect.

A number of factors contribute to the formation of the urban heat island effect in cities. The primary driver of the urban heat island is the loss of canopy cover resulting from urbanization (Heinl, Hammerle, Tappeiner, & Leitinger, 2015). With the loss of vegetation, there is less shading and evaporative cooling. Second, during the day solar energy heats up impervious surfaces, which store the solar energy as heat energy (known as thermal capacity). This heat energy is radiated back into the environment especially during the cooler periods of night. Third, the geometry of cities—the density of buildings, percentage of impervious cover, and amount of reflective surfaces—affects not only the amount of heat energy stored but also the air movement through the city. In densely packed cities, air movement can be restricted, which often eliminates cooling breezes and traps hot area between buildings (see EPA, 2008). .

The urban heat island effect significantly affects atmospheric processes in cities. For example, it alters precipitation patterns and ozone formation. Cities can create their own weather and affect the weather of adjacent areas (Shepherd, 2005). Because of the urban heat island and other factors such as dust, moisture, and city geometry, areas downwind of a city can receive higher rainfall than historically measured. In fact, as a city grows the rain pattern can shift from adjacent areas to the city itself. This shift in precipitation creates localized droughts affecting not only native communities but also adjacent agricultural lands (Shepherd, 2005). Other changes in meteorological events include reduction in wind speeds, altered radiation, and lower humidity (Sukopp, 2004).

Similarly, the urban heat island effect increases not only ozone production but also number of days of unhealthy ozone levels in urban areas as well as areas downwind of cities. Ozone is a photochemical reaction between volatile organic compounds (VOCs) and nitrogen oxides. Formation primarily occurs during the hot days of summer. Although VOCs are produced naturally, man-made VOCs (e.g., gasoline and other petroleum products) are the principal source for ozone formation.

Plants are especially vulnerable to high levels of ozone, which affects leaf tissue and the plant's ability to grow. Although ozone damages plant tissues, the net effect of multi-gas interactions that are common in the urban atmosphere on natural communities is relatively unknown (Kaye, Groffman, Grimm, Baker, & Pouyat, 2006). Finally, the high surface temperatures of impervious surfaces (e.g., roadways) transfer significant amounts of heat to storm water runoff, which often flows into streams and other water bodies. This thermal pollution can increase stream temperatures by as much as 15°C in a relatively short period of time, resulting in significant impacts on aquatic life, both plants and animals (EPA, 2008).

Although higher day and nighttime temperatures create a significant heat load for both humans and native species, a number of native and non-native species actually thrive in these conditions. Both poison ivy (*Rhus radicans*) and ragweed (*Ambrosia artemisiifolia*) have shown a significant increase in growth in urban as compared with rural environments because of higher temperatures and CO_2 levels (Ziska, Epstein, & Rogers, 2008). The higher levels of CO_2 result from the higher combustion of fossil fuels in cities than in rural areas. These and other species can have a significant effect on human health as well as altering natural communities, especially as cities get warmer.

Ambient temperatures in cities are expected to rise with climate change. The combination of the urban heat island and the increase in global temperatures can cause extreme temperature events within cities. For instance, in Shanghai, China, human mortality has increased significantly because of the combination of the effect of the urban heat island and heat waves (Tan et al., 2010). Similar synergistic effects have been reported for Germany, France, and Italy.

Invasive Species and Novel Ecosystems

Cities are currently the epicenter for the introduction of non-native invasive species, both plants and animals (Reichard & White, 2001). An invasive species is often a non-native species that has been either deliberately or inadvertently introduced into a new environment. They can spread rapidly to other locations and become economically and biologically harmful. Examples of deliberate introductions include pets such as Boa constrictors, and nursery and garden plants such as purple loosestrife (*Lythrum salicaria*) and ice plant (*Carpobrotus edulis*). Examples of inadvertent introductions include zebra mussel (*Dreissena polymorpha*) from ballast water from ships docking at urban ports, Spanish slug (*Arion vulgaris*) from horticultural plants (Kowarik, 2011), and Asian longhorn beetle (*Anoplophora glabripennis*) from packaging materials. Invasive species cause over $137 billion, annually, in the United States and 12 billion euros in the European Union in economic losses and are second only to habitat destruction for negatively effecting the viability of native species (Pimentel, Lach, Zuniga, & Morrison, 2000; Sundseth, 2016).

Domestic cats (*Felis catus*) and dogs (*Canis lupus familiaris*) are invasive species. Cats and dogs, especially feral cats and dogs, can have a tremendous effect on native biodiversity. Cats kill an estimated 470 million birds per year in the United States (Pimentel, 2007). In addition, cats kill small mammals, amphibians, and reptiles. Dogs can also kill native species such as deer, rabbits, and small mammals, but they primarily kill livestock.

A counter-perspective to the problems caused by invasive species is that they can contribute significantly to the ecosystem services enjoyed by urban residents. For example, Tree of Heaven (*Ailanthus altissima*) makes up 25% of the leaf area in Brooklyn, New York (Nowak, Crane, Stevens, & Ibarra, 2002). This species requires no management activities and its removal would have a significant effect on the city's environment in terms of lost rain interception and shading. Similarly, some non-native species may be more adaptive to xeric conditions in desert cities than native species (Pataki, McCarthy, Litvak, & Pincetl, 2011). In the Los Angeles basin, native tree species are often riparian species and may require more water to grow than a non-native tree species adapted to a drier environment. Nevertheless, one cannot condone planting invasive species for personal enjoyment, beautification, or enhancement of ecosystem services in cities. It is necessary to be cognitive of the impacts of these species within cities and beyond their boundaries. Selection of non-native species for horticultural and management purposes needs to follow guidelines and recommendations to minimize invasive species introductions (Reichard & White, 2001).

Urbanization also creates novel ecosystems. A novel ecosystem is a community of species that is unique and does not reflect species composition or structure of past or present communities (Hobbs, Higgs, & Higgs, 2009). Generally, these ecosystems occur on disturbed sites and contain a mixture of both native and non-natives species. Novel ecosystems are frequently associated with urban and urbanizing landscape because of the disturbances created through land conversions and land abandonment, new climatic conditions (e.g., urban heat island), and the high abundance of non-native species, some of which are invasive (Kowarik, 2011). The resulting novel community can impede native species regeneration and alter soil processes. For example, in upstate New York remnant urban forest patches are dominated by *Quercus* spp. (Oak species) and *Acer saccharum* (sugar maple). On the other hand, emergent forest patches (newly developed) are dominated by *A. negundro* (box elder) and *A. platanoides* (Norway maple), an invasive species (Zipperer, 2002). Because of the dominance of *A. platanoides*, these emergent patches are novel when compared with native communities found in the region. Street trees provided the seed source for *A. platanoides*. Other examples of novel communities include gardens, vacant lots, and wastelands. Nonetheless, these novel communities can provide ecosystem services to humans and may have conservation value to native species (Kowarik, 2011).

Urban Soils

Urbanization affects soils, specifically biogeochemical processes, which in turn affect community productivity and growth. Soil biogeochemistry is defined as the study of physical, chemical, and biological processes that occur in soil (Kaye et al., 2006). Important biogeochemical processes include the nitrogen and carbon cycles. Likewise, soils serve as sinks for heavy metals and are important with respect to water infiltration. So, what is an urban soil? Urban soils are modified soils often containing elements of the urban environment (broken bricks, glass, ashes, crushed stone, concrete) and natural or earth materials (Effland & Pouyat, 1997). They may be toxic, low in nutrients, and highly compressed.

Because of the occurrence of concrete, urban soils are often alkaline rather than acidic like “natural” soils. This change in pH affects the biogeochemical processes that govern nutrient cycling and directly influences community composition and productivity.

The urban environment, with its air, soil, and water pollution, CO_2 emissions, urban heat island, and introduction of non-native species, has drastically affected biogeochemical processes (Kaye et al., 2006). It needs to be pointed out that these factors do not act independently. In fact, they often act synergistically on natural systems. For instance in New York City, the combination of higher soil temperatures from the urban heat island effect, nitrogen deposition through air pollution, and the presence of an invasive earthworm (*Lumbricus rebellus*) have altered nitrogen cycling in temperate urban woodlands (Steinberg, Pouyat, Parmlee, & Groffman, 1997). The presence of earthworms increases rates of mineralization and denitrification, which affects the amount of nitrogen available to plants and microorganisms.

Urban areas have higher concentrations of atmospheric pollutants such as nitrate oxides, sulfur oxides, and heavy metals than rural areas because of fossil fuel combustion and industrial emissions (Pouyat et al., 2007). These pollutants result in higher amounts of nitrogen, alkaline dust particles, and heavy metals entering urban soils through wet and dry deposition, which can affect plant growth, rates of decomposition, and nutrient processes. Similarly, biogeochemical characteristics of urban soils are affected by human activities such as irrigation and fertilizing. These activities create artificial environments that alter nutrient cycling and availability, affect water quality, and favor a suite of species (often non-native and potentially invasive) adapted to living in urban landscapes. Ultimately, it is these activities and the built infrastructure that are the primary drivers that affect natural communities and ecosystems services.

Even though urbanization directly and indirectly affects terrestrial systems, proper planning can minimize these effects. As seen with hydrologic systems, restoration and planning can create opportunities to maintain or improve not only native biodiversity but also urban sustainability itself. For instance, maintaining large natural habitats as well as smaller areas can preserve connectivity and unique habitats, thus conserving biodiversity (McDonald, Güneralp, Zipperer, & Marcotullio, 2014). Similarly, restoring degraded forest habitats can enable native species to regenerate and grow (Johnson & Handel, 2016). Furthermore, planting gardens and allowing fields to fallow can contribute habitat for native species, especially insects and other arthropods. Overall, it is important to include ecological and environmental structure and attributes as part of land-use decisions and recognize that these attributes contribute to the services and benefits natural systems provide to humans.

Conclusion: Urban Planning and Sustainability

As of 2020, more than 50% of the world’s population lives in cities. By 2050 this percentage will exceed 60%, creating an extraordinary demand for natural resources and placing exceptional stress on natural systems. How can a healthy economy be maintained without depleting natural resources? How can cities develop and grow while maintaining a healthy environment for future generations? There are no easy answers to questions such as these, but they must be dealt with by utilizing the best information available (National Academies of Science, 2016).

The growth of most cities worldwide is not sustainable because of the consumption and alteration of natural systems. There are no set rules or templates for cities to grow ecologically, economically, or socially sustainably. Each city, because of its governance, environment, economy, and social structure and well-being, has a unique set of challenges and opportunities. At the same time, cities do share common challenges such as poverty, transportation, and social inequities. Based on uniqueness and commonalities, recommendations have been developed to guide cities towards a more sustainable pattern of environmental, economic, and social development for the future (National Academies of Science, 2016; Table 2). How cities implement these recommendations will ultimately depend upon prevailing societal values and an appreciation for the intimate relationship between human well-being and the ecological systems that support us all.

To aid in planning, non-governmental organizations (NGOs) such as the International Council for Local Environmental Initiatives (ICLEI) work with local planners and decision-makers to make cities and communities more sustainable—creating urban landscapes for human needs while conserving biodiversity. Planning needs to be done not only locally but also regionally and nationally, thus linking practices and management. Likewise, citizens are forming citizen science and monitoring groups to track how their environment is changing with urbanization, becoming involved in the decision-making process to address problems, and developing plans to improve their environment. In fact, long-term monitoring of the environment, biodiversity, and natural ecosystems (as well as social ecosystems) is paramount to understand how the environment, biodiversity, and these systems are changing through time and how they are being influenced by urbanization and climate change. By knowing how systems are changing, cities can be adaptive and manage the urban landscape to maintain or enhance ecosystem services.

For instance, depleted and degraded water resources associated with past urban infrastructure is driving communities to look for new systems for managing storm water. These systems are recycling storm water, harvesting rainwater, slowing down peak storm water flows, and significantly reducing or eliminating discharges to receiving waters from both pipes and land uses (Visser, Moran, Singleton, & Esser, 2018). These “new hybrid” systems (also known as green infrastructure) transcend natural and built environments within communities to reconfigure and restore hydrological and ecological functions simultaneously. They provide water to meet the needs of the community, and protect the health of people and the native communities that sustain biodiversity.

Table 2. Areas of Focus to Help Guide Cities within the United States to Be More Sustainable (modified from National Academies of Science, 2016)

Area of focus	**Reasons**
Being a good neighbor	A city is part of a larger regional context and needs to recognize that context when planning so as to not affect negatively other cities.
Multiple scales	Environmental, economic, and social processes operate across multiple scales. These multiple linkages and feedbacks need to be integrated into the decision-making process.
Socio-ecological systems	Cities are socio-ecological systems, and identifying policies that create synergy among linkages and feedback can result in multiple co-benefits for the environment, economy, and society.
Sharing	Cities can learn from each other. Share experiences to identify working policies. See ICLEI.
Importance of science	Science-based decision and planning. Metrics, used to measure success, should be based on research.
Partners, Partners, Partners	Sustainability is only achieved through partnerships within and across governmental agencies, communities, and individual groups as well as other cities. The city, collectively, is deciding its future.
Long-term planning	Policy decisions affecting long-term sustainability must be built into the governance itself so that the plans and policies are not derailed with each election cycle.
Reducing inequities	For a city to be sustainable, inequities among social groups need to be addressed across the range of economic, social, and cultural attributes.

Performance standards	Based on science, benchmarks and performance standards enable cities to be adaptive in addressing successes or deficiencies in policy decisions to attain sustainability goals.
Planning Tomorrow, Today	Often, when a natural resource is depleted or destroyed, it is gone forever. To reduce future losses, planning must begin today! Both short-term and long-term planning are needed to conserve and protect natural resources in the long term. Environmental, economic, and social policies must complement and not degrade or diminish each other.

With population growth, growing metropolitan regions expand their influence on the use and condition of an ever greater portion of the Earth’s surface. Biodiversity and natural communities within and outside municipal boundaries are affected by this growth. Designing urban landscapes to maintain biodiversity and natural communities is paramount. Designing with nature is not a new idea (McHarg, 1969). In fact, conservation planning is occurring across multiple disciplines. For instance, engineering professionals have expanded their paradigms by incorporating a more ecological perspective into their work (Mitsch & Jørgensen, 2003). Similarly, landscape architects and designers are embracing conservation planning in their site plans and site development (Ahern, Leduc, & York, 2006). For example, communities in Berlin, Germany, are transforming their lawns into “wild” yards where native species colonize sites and create “natural” areas for aesthetics and recreational usage (Ignatieva & Hedblom, 2018). These natural areas increase species diversity not only for plants but also for insects and other arthropods that use the sites for food, nesting, and security. Nonetheless, more conservation planning is needed to conserve biodiversity and natural communities in metropolitan regions (McDonald et al., 2014).

The bottom line for cities is that mitigating environmental degradation and native biodiversity losses because of urbanization must be addressed with the same intensity as economic and social issues. Without native biodiversity, the ecosystem services that humans depend on will be severely diminished, degraded, or even lost. The environmental effects of urbanization can be offset through proper planning and decision-making if humans make that choice.

Further Reading

Müller, N., Ignatieva, M., Nilon, C. H., Werner, P., & Zipperer, W. C. (2013). Patterns and trends in urban biodiversity and landscape design. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, … C. Wildkinson (Eds.), *Urbanization, biodiversity, and ecosystem services: Challenges and opportunities* (pp. 123–174). New York, NY: Springer.

date: 01 September 2020

Niemelä, J. (Ed.). (2011). *Urban ecology: Patterns, processes, and applications*. Oxford, UK: Oxford University Press.

Pickett, S. T. A., Cadenasso, M. L., Grove, J. M., Nilon, C. H., Pouyat, P. V., Zipperer, W. C., & Costanza, R. (2001). Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. *Annual Review of Ecology and Systematics*, *32*, 127–157.

References

Ahern, J., Leduc, E., & York, M. L. (2006). *Biodiversity planning and design*. Washington, DC: Island Press.

Arnold, C. L., Jr., & Gibbons, C. J. (1996). Impervious surface coverage: The emergence of a key environmental indicator. *American Planning Association Journal*, *62*, 243–258.

Bernhadrdt, E. S., & Palmer, M. A. (2007). Restoring streams in an urbanizing world. *Freshwater Biology*, *52*, 738–751.

Biswas, A. K., Tortajada, C., & Stavenjagen, M. (2018, June 2). **In an urbanizing world, shrinking cities are a forgotten problem**. *The Conservation*.

DeFries, R. S., Rudel, T., Uriarte, M., & Hansen, M. (2010). Deforestation driven by urban population growth and agricultural in the twenty-first century. *Nature Geoscience*, *3*, 178–181.

Effland, W. R., & Pouyat, R. V. (1997). The genesis, classification, and mapping of soils in urban areas. *Urban Ecosystems*, *1*, 217–228.

EPA. (2008). *Reducing urban heat islands: Compendium of strategies*. Washington, DC: Environmental Protection Agency.

Fischer, J. D., Schneider, S. C., Ahlers, A. A., & Miller, J. R. (2015). Categorizing wildlife responses to urbanization and conservation implications of terminology. *Conservation Biology*, *29*, 1246–1248.

Forman, R. T. T., & Alexander, L. E. (1998). Roads and their major ecological effects. *Annual Review of Ecology and Systematics*, *29*, 207–231.

Forum, W. E. (2013). *The Green Investment Report: The ways and means to unlock private finance for green growth*. Geneva, Switzerland.

Groffman, P. M., Bains, J. B., Band, L. E., Belt, K. T., Brush, G. S., Grove, J. M., … Zipperer, W. C. (2003). Down by the riverside: Urban riparian ecology. *Frontiers in Ecology and the Environment*, *1*, 315–321.

Heinl, M., Hammerle, A., Tappeiner, U., & Leitinger, G. (2015). Determinants of urban–rural surface temperature differences: A landscape scale perspective. *Landscape and Urban Planning*, *134*, 33–42.

Hobbs, R. S., Higgs, E., & Higgs, J. A. (2009). Novel ecosystems: Implications for conservation and restoration. *Trends in Ecology and Evolution*, *24*(11), 599–605.

Ignatieva, M., & Hedblom, M. (2018). An alternative urban green carpet. *Science*, *362*, 148–149.

Johnson, L. R., & Handel, S. N. (2016). Restoration treatments in urban park forests drive long-term changes in vegetation trajectories. *Ecological Applications*, *26*(3), 940–956.

Joo, Y., & Seo, B. (2018). Dual policy to fight urban shrinkage: Daegu, South Korea. *Cities*, *73*, 128–137.

Kaye, J. P., Groffman, P. M., Grimm, N. B., Baker, L. A., & Pouyat, R. V. (2006). A distinct urban biogeochemistry. *Trends in Ecology and Evolution*, *21*, 192–199.

Knoema. (2020). **World Data Atlas: Germany**.

Kolpin, D. W., Furlong, E. T., Meyers, M. T., Thurman, E. M., Zaugg, S. D., Barber, L. B., & Buxton, H. T. (2002). Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. streams, 1999–2000: A national reconnaissance. *Environmental Science and Technology*, *36*, 1202–1211.

Kowarik, I. (2011). Novel urban ecosystems, biodiversity and conservation. *Environmental Pollution*, *159*, 1974–1983.

Levenson, J. B. (1981). Woodlots as biogeographic islands in Southeastern Wisconsin. In R. L. Burgess & D. M. Sharpe (Eds.), *Forest island dynamics in man-dominated landscapes* (pp. 13–40). New York, NY: Springer-Verlag.

Lewis, J. A., Zipperer, W. C., Ernstson, H., Bernik, B., Hazen, R., Elmqvist, T., & Blum, M. J. (2017). **Disparate socioecological responses to disturbance in New Orleans following Hurricane Katrina**. *Ecoshpere*, *8*(9), 1–24.

Loss, S. R., Will, T., Loss, S. S., & Marra, P. P. (2014). Bird-building collisions in the United States: Estimates of annual mortality and species vulnerability. *The Condor: Ornithological Applications*, *116*, 8–23.

Mallach, A., Haase, A., & Hattori, K. (2017). The shrinking city in comparative perspectives: Contrasting dynamics and responses to urban shrinkage. *Cities*, *69*, 102–108.

McDonald, R., Güneralp, B., Zipperer, W., & Marcotullio, P. (2014). The future of global urbanization and the environment. *Solutions*, *6*, 60–69.

McHarg, I. L. (1969). *Design with nature*. Garden City, NY: Doubleday/Natural History Press.

Milanski, B. (2019). The impact of migration on Europe. *Journal of European and Balkan Perspectives*, *2*(1), 8–16.

Mitsch, W. J., & Jørgensen, S. E. (2003). Ecological engineering: A field whose time has come. *Ecological Engineering*, *20*(5), 363–377.

National Academies of Science. (2016). *Pathways to urban sustainability: Challenges and opportunities for the United States*. Washington, DC: National Academies Press.

Nelson, A. C. (2004). *Towards a new metropolis: The opportunity to rebuild America*. Washington, DC: The Brookings Institute.

Nowak, D. J., Crane, D. E., Stevens, J. C., & Ibarra, M. (2002). *Brooklyn's urban forest*. Newtown Square, PA: USDA Forest Service GTR NE-290.

Pataki, D. E., McCarthy, H. R., Litvak, E., & Pincetl, S. (2011). Transpiration of urban forests in the Los Angeles metropolitan area. *Ecological Applications*, *21*(3), 661–677.

Paul, M. J., & Meyer, J. L. (2001). Streams in the urban landscape. *Annual Review of Ecology and Systematics*, *32*, 333–365.

Pimentel, D. (2007). **Environmental and economic costs of vertebrate species invasions into the United States**. *Managing Vertebrate Invasive Species*, *8*.

Pimentel, D., Lach, L., Zuniga, R., & Morrison, D. (2000). Environmental and economic costs of nonindigenous species in the United States. *Bioscience*, *50*(1), 53–65.

Pouyat, R. V., Pataki, D. E., Belt, K. T., Groffman, P. M., Hom, J., & Band, L. E. (2007). Effects of urban land-use change on biogeochemical cycles. In J. G. Canadell, D. E. Pataki, & L. F. Pitelka (Eds.), *Terrestrial ecosystems in a changing world* (pp. 45–58). New York, NY: Springer.

Reichard, S. H., & White, P. S. (2001). Horticulture as a pathway of invasive plant introductions in the United States. *Bioscience*, *51*, 103–113.

Robinson, S. K., Thompson, F. R., III, Donovan, T. M., Whitehead, D. R., & Faaborg, J. (1995). Regional forest fragmentation and nesting success of migratory birds. *Science*, *267*, 1987–1990.

Rosi, E. J., Snow, B. D., Rojas, M., Rohas, A. J., Reisinger, A. J., & Kelly, J. J. (2018). Urban stream microbial communities show resistance to pharmaceutical exposure. *Ecosphere*, *9*(1), 1–16.

Santos, T., & Telleria, J. L. (1994). Influence of forest fragmentation on seed consumption and dispersal of Spanish juniper *Juniperus thurifera*. *Biological Conservation*, *70*(2), 129–134.

Seto, K., Güneralp, B., & Hutra, L. R. (2012). Global forecast of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proceedings of the National Academy of Science*, *109*, 16083–16088.

Shepherd, J. M. (2005). A review of current investigations of urban-induced rainfall and recommendations for the future. *Earth Interactions*, *9*, 1–27.

Steinberg, D. A., Pouyat, R. V., Parmlee, R. W., & Groffman, P. M. (1997). Earthworm abundance and nitrogen mineralization rates along urban–rural land use gradient. *Soil Biology and Biogeochemistry*, *29*, 427–430.

Sukopp, H. (2004). Human-caused impact on preserved vegetation. *Landscape and Urban Planning*, *68*, 347–355.

Sun, G., & Lockaby, B. G. (2012). Water quantity and quality at the urban–rural interface. In D. A. Laband, B. G. Lockab, & W. C. Zipperer (Eds.), *Urban–rural interface* (pp. 29–48). Madison, WI: Soil Science Society of America.

Sundseth, K. (2016). *Invasive alien species: A European Union response*. Brussels, Belgium: European Union.

Tan, J., Zheng, Y., Tang, X., Guo, C., Li, L., Song, G., … Chen, H. (2010). The urban heat island and its impact on heat waves and human health in Shanghai. *International Journal of Biometeorology*, *54*, 75–84.

United Nations. (2014). *World urbanization prospects: The 2014 revision, highlights*. New York: Department of Economics and Social Affairs, Population Division.

United Nations. (2016). *The world’s cities in 2016—Data booklet*. New York: Department of Economics and Social Affairs, Population Division.

Visser, A., Moran, J. E., Singleton, M. J., & Esser, B. K. (2018). Importance of river water recharge to the San Joaquin Valley groundwater system. *Hydrological Processes*, *32*(9), 1202–1213.

von der Lippe, M., & Kowarik, I. (2008). Do cities export diversity? Traffic as dispersal vector across urban–rural gradients. *Diversity and Distribution*, *14*, 18–25.

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan, R. P., II. (2005). The urban stream syndrome: Current knowledge and the search for a cure. *Journal of the North American Benthological Society*, *24*(3), 706–723.

Zipperer, W. C. (1993). Deforestation patterns and their effects on forest patches. *Landscape Ecology*, *8*, 177–184.

Zipperer, W. C. (2002). Species composition and structure of regenerated and remnant forest patches within an urban landscape. *Urban Ecosystems*, *6*, 271–290.

Ziska, L. H., Epstein, P. R., & Rogers, C. A. (2008). Climate change, aerobiology, and public health in the Northeast United States. *Mitigation and Adaptation Strategies for Global Change*, *13*, 607–613.

Wayne C. Zipperer

USDA Forest Service - Southern Research Station 2306 Mowry Rd Gainesville Florida 32611 United States

Robert Northrop

University of Florida IFAS Extension-Hillsbourgh C. Seffner, Florida United States

Michael Andreu

University of Florida - School of Forest Resources and Conservation Gainesville, Florida United States