CHAPTER 10

Climate change and urban forest soils

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ABSTRACT

With the proportion of the global population living in urban areas expected to reach 67% by 2040, soils located in these areas, or urban soils, will become increasingly important, particularly with regards to how these soils respond to climate change and their ability to provide ecosystem services. Moreover, urban areas to a large degree have already experienced elevated air temperatures, carbon dioxide concentrations, and increased variability in precipitation, and thus can serve as analogs for how forest soils will respond to climate change. This chapter reviews the wide range of soil characteristics in urban landscapes, their response to both urban environments and changes in climate, their potential to serve as analogs for understanding the future effects of climate change on soils, and how they provide ecosystem services including those that assist in mitigating the effects of greenhouse gas emissions.

Introduction

Urban populations are increasing with the number of mega (10 million people or more) sized cities expected to increase from 10 in 1990 to 41 in 2030. Additionally, 67% of the global population is expected to reside in urban and peri-urban areas by 2040 (United Nations, 2015). In the United States, the human population is not growing as fast as the rest of the world, but proportionally the expansion rate of urban areas has kept pace, or exceeded, global estimates with land converted to urban uses growing by more than 34% between 1980 and 2000 (USDA Natural Resources Conservation Service, 2001). The current estimate of urban land area is almost 6% of the total conterminous United States (Homer et al., 2015). Additionally, in the latest national census almost 250 million people lived in urban and suburban areas, which roughly accounts for 81% of the total population of the United States (United States Census Bureau, 2010).

The conversion of land from primarily agricultural and forest uses to urbanized landscapes has the potential to greatly modify the Nation's soils especially with respect to their ability to store carbon (C) and mitigate the release of greenhouse gases (Trammell et al., 2018). Additionally, changes in climate, including higher temperature and more intense precipitation events, may in turn affect the potential of soils in urbanized areas, or urban soils, to provide various ecosystem services, some of which are vital for reducing the vulnerability of densely populated areas to natural disasters, such as flooding (Anne et al., 2018).

The impacts on soil in the conversion of native ecosystems to agricultural use, and recovery from agricultural use, have been relatively well studied, while conversions to urban land uses have received little attention, particularly with respect to climate and environmental changes. We know, for example, that as agricultural practices have been abandoned in previously forested areas in the United States,

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forest regrowth in these soils has resulted in a gradual recovery of aboveground, and to a lesser extent belowground, C pools over many decades (Caspersen et al., 2000; Flinn and Marks, 2007). With urban land conversions, soil scientists are just beginning to understand how soils respond to, let alone recover, from the long-term effects of urban land use, especially with respect to a rapidly changing climate.

The purpose of this chapter is to explore the array of modifications of soil from anthropogenic activities in urban areas, particularly within the context of global environmental change. Specifically, we: (1) define and describe urban soils and their physical, chemical and biological characteristics; (2) predict urban soil response to climate change, particularly in remnant forest patches; and (3) consider the potential of urban soil to provide ecosystem services and mitigate future climate change.

What is an urban soil?

The term "urban soil" was first used over 50 years ago by Zemlyanitskiy (1963) to describe the characteristics of soils located in urban areas. Urban soil was defined by Craul (1992) as "a soil material having a non-agricultural, man-made surface layer more than 50 cm thick that has been produced by mixing, filling, or by contamination of land surface in urban and suburban areas." This definition was in part derived from earlier definitions by Bockheim (1974) and Craul and Klein (1980), which similarly focused on highly modified soils. As soil scientists took a broader view of urban landscapes and human impacts on soil in general, they used the term "anthropogenic soil", which placed urban soils in a broader context of human altered soils rather than limiting the definition to urban and suburban areas alone (Evans et al., 2000; Capra et al., 2015). To recognize even a wider set of observed soil conditions in urban landscapes, Effland and Pouyat (1997), Lehmann and Stahr (2007), and Morel et al., (2017) more broadly defined urban soils to include soils that are relatively undisturbed yet altered by urban environmental changes, such as the deposition of atmospheric pollutants or warmer air temperatures. This broader view of urban soil results in a continuum of soil conditions occurring in an urban landscape at any point in time, which is the definition used in this chapter.

The Urban Soil Continuum

Based on the progression of efforts seeking to understand and define urban soils, it has become obvious to scientists that the range of soil conditions that occur in urban and suburban landscapes varies widely — perhaps more widely than those encountered in rural landscapes (see Morel et al., 2017). Indeed, soils in urban areas have been found to range widely with respect to all variables used to characterize soils, which can be best described as a continuum of conditions, or the "Urban Soil Continuum" (USC). The USC includes the full range of urban soil types, from types with relatively low human impacts (e.g., remnant native forest or grassland soil) to those that are greatly impacted by human activities that typically occur in urban areas (Fig. 10.1). The least modified soil types have not been physically disturbed, but rather are primarily affected by environmental changes associated with urban land uses. The most highly modified soils are without structure (Short et al., 1986), made up of human-transported materials (Shaw and Isleib, 2017), sealed at the surface (Scalenghe and Marsan, 2009), highly managed as with public or residential lawns (Trammell et al. 2016), or engineered for specific purposes such as tree pits (Grabosky et al., 2002) and rain gardens

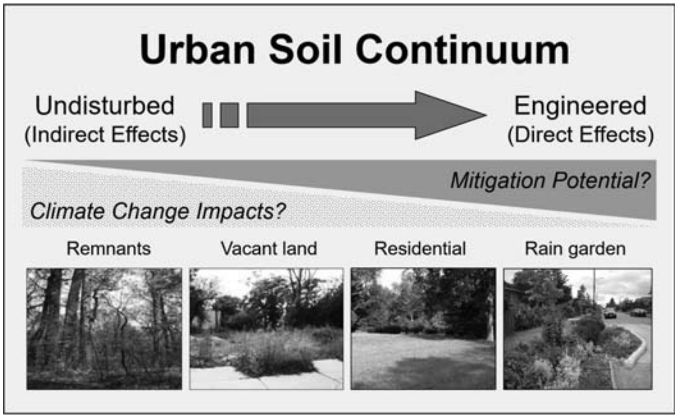

FIG. 10.1

A conceptual characterization of the Urban Soil Continuum (USC). Soil conditions within the USC range from the least impacted soils that are associated with remnant patches of forest or grassland (left side of continuum) to highly disturbed and managed soils that are associated with developed areas (right side of continuum). The most modified soils include those that are engineered to provide various ecosystem services. It is hypothesized that soils being effected by urban environmental changes (indirect effects) will be the most sensitive to climate change, while the most managed or engineered soils have the most potential to mitigate the effects of climate change, if appropriately designed.

(Ahiablame et al., 2012). Intermediate in modification are soil types that are associated with abandoned land or are managed, but experience relatively low impacts from ongoing disturbance such as perennial gardens (Edmondson et al., 2014). Where soil types fall along this continuum should reflect how they will respond to long-term changes in climate and their ability to mitigate greenhouse gas emissions or provide ecosystem services.

Direct and indirect effects

Soils become modified in urban areas through direct or indirect effects and often times both (Fig. 10.2). Direct effects include those generally associated with soil modifications occurring on the more highly disturbed end of the USC (Fig. 10.1). For most urban areas, these types of impacts occur during, rather than after, the land development process including leveling and moving soils with heavy equipment, excavating and refilling, or contouring and creating urban landscape features. Soil management practices such as fertilization and irrigation that are introduced post development are also considered direct effects. Management is generally an effort by humans to overcome some limitation frequently caused by a direct effect and often result in highly productive soils (Pouyat et al., 2007).

Whereas direct effects largely occur from human disturbances, indirect effects involve changes in the abiotic and biotic environment that are caused by human activities (e.g., emission of pollutants), which occur in some form across the entire USC, but with a proportionately greater effect on undisturbed soils associated with remnant parcels (Fig. 10.1). Urban environmental factors include the urban heat island (Mount et al., 1999; Imhoff et al., 2010; Savva et al., 2010; Hall et al., 2015),

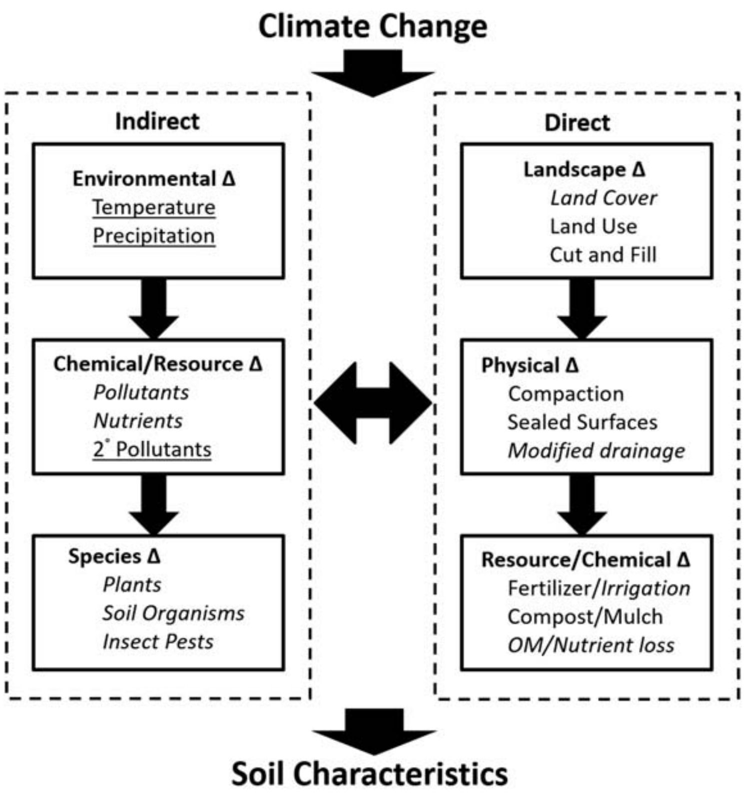

FIG. 10.2

Conceptual model showing the indirect and direct effects of urban land uses on soil characteristics. Indirect effects (left column) result in environmental, chemical, and species changes, while direct effects (right column) result in landscape, soil physical, and resource availability changes. Both indirect and direct factors may interact. Factors expected to be the most sensitive to climate change are underlined and those of intermediate sensitivity are in italics.

introductions of exotic plant and animal species (Steinberg et al., 1997; Ehrenfeld et al., 2001; Lilleskov et al., 2010; Szlavecz et al., 2011), and atmospheric deposition of pollutants such as nitrogen (N) and sulfur (S) (Lovett et al., 2000; Juknys et al., 2007; Rao et al., 2014; Huang et al., 2015; Decina et al., 2017), heavy metals (Yu et al., 2012; Datko-Williams et al., 2014; Werkenthin et al., 2014), and potentially toxic organic chemicals (Wong et al., 2006; Zhang, 2006; Jensen et al., 2007). For all of these urban environmental factors, climate change will differentially influence the magnitude of their effects (Fig. 10.2).

Response of urban soils to climate change

Changes to the global climate due to human activity already occur to a large degree in urban areas (Carreiro and Tripler, 2005). Air temperatures and atmospheric concentrations of carbon dioxide

(CO_2) and methane (CH_4) are up to 30% and 40% higher in urban compared to rural or global averages, respectively (Ziska et al., 2003; Altshullera et al., 1996). These environmental changes represent the overall increases for atmospheric CO_2 expected at a global scale in 30 or more years (Meinshausen et al., 2011). How expected increases in air temperature (especially at night), rainfall intensity, and atmospheric concentrations of CO_2 resulting from changes in global climate will affect urban soils can best be described through their influences on the direct or indirect effects of urban land use discussed earlier. Indirect effects can be associated with environmental, chemical, and species changes, while direct effects include those associated with landscape, physical and resource changes that occur during and post-urban development (Fig. 10.2). Furthermore, the least impacted soils along the USC, such as remnant forest soils, are most influenced by indirect effects and thus should be the most impacted by long-term changes in global climate, whereas for soils that are managed or more impacted by direct effects, climate change would be expected to be less influential (Fig. 10.1).

Indirect effects

Environmental change

One of the more important effects of urban environments on soil properties is the modification of soil temperature and moisture regimes (Pouyat et al., 2007, Fig. 10.2). Urban environments are characterized by a localized increase in temperature known as the "heat island" effect (Oke, 1990). Urban heat islands (UHI) are caused by both a reduction in evapotranspiration and an increase in heat storage by urban structures (Oke, 1990). These changes in the flux of latent heat increase both the maximum and minimum temperatures of urban environments. Temperature differences between the urban core and outlying rural areas generally range from 1 to 3 °C annually, yet can be as high as 12 °C on occasion, and usually are greatest at night (Forman, 2014). A rise in average minimum temperature has increased the number of frost-free days in urban areas, which extends the growing season for urban plants relative to the surrounding countryside. Accordingly, the UHI is linked with changes in plant phenology, including timing and duration of canopy leaf out, leaf budburst and flowering (Jochner and Menzel, 2015; Chen et al., 2016). In combination with the UHI effect, elevated atmospheric CO_2 concentrations may delay autumn leaf fall (Taylor et al., 2008) lengthening the time of urban tree growth.

There are few data addressing whether these differences in air temperature are also reflected in the temperature of urban soils. Mean annual temperatures in highly disturbed soils (0−10 cm depth) on a playground in New York's Central Park were more than 3 °C warmer than in soils in an adjacent wooded area (Mount et al., 1999). Savva et al. (2010) found that average annual soil temperature was higher in turfgrass than in forest remnant soils in the metropolitan area of Baltimore, USA (15.0 vs. 12.6 °C). In comparing soil temperatures in the Urbana-Champaign, USA area, monthly mean temperature at a depth of 10 cm was 4.1 °C higher in the disturbed urban locations than in a nearby forest stand (Graves, 1994).

While the causal drivers of the UHI is largely understood, the alteration of precipitation in cities appears to be more complex (Song et al., 2016). A few studies have identified an "urban rainfall effect" (Shepherd and Burian, 2003), where urban areas experience increased rainfall, snowfall, and convection storm events compared to nearby rural areas (Shem and Shepherd, 2009; Niyogi et al., 2011). The combined effect of altered temperature and precipitation regimes in turn can affect plant productivity, detrital inputs of organic matter, and microbial activity, all of which have the potential to influence urban soil characteristics. Additionally, greater intensity of rainfall may increase water availability for plants and soil organisms in excessively-drained soils; however, it is more likely that more intense rainfall events will lead to greater stormwater runoff and thus erosion (National Research Council, 2009). Overall, in urban areas the impacts of climate change on both soil temperature and moisture regimes is expected to be pronounced given the environmental changes already occurring in urban areas (Fig. 10.2).

Chemical/resource change

Urban environments are typically altered via inputs of various chemicals through emissions of gases and wet and dry deposition (Fig. 10.2). In urban areas, atmospheric concentrations of CO_2 can reach as much as 520 ppm (Koerner and Klopatek, 2002; Pataki et al., 2003), or nearly double the preindustrial level of 280 ppm. The elevated level of CO_2, in addition to enhanced deposition of inorganic nitrogen (N), may actually benefit plant and soil microbial productivity in urban areas (Ziska et al., 2004). On the other hand, the occurrence of trace metals and other contaminates in urban deposition may inhibit plant and microbial productivity (Bååth, 1989; Gill 2014). Additionally, urban environments experience from the combustion of fossil fuels elevated nitrogen oxide (NO_x) emissions, which along with carbon monoxide and volatile organic compounds can react in the troposphere to form ozone (O_3), which is a strong oxidant that can severely impact plant health (Gregg et al., 2003). Overall, the impacts of climate change on pollutant emissions in urban areas should be significant, especially for pollutants formed from secondary reactions such as O_3. For example, ozone production rises during heatwaves, because plants absorb less ozone (Emberson et al., 2013).

Species change

Finally, and perhaps the least obvious, are plant and soil faunal species changes that are often associated with urban land uses (Fig. 10.2). Depending on the taxa, the conversion of native habitats to urban land uses generally leads to a loss of native species; however, in many cases native species continue to persist in urban landscapes (Kowarik, 2011). As native habitats are converted to urban uses, humans both purposely and accidently introduce non-native species such that measures of biodiversity are often much higher in urban versus rural areas for a particular metropolitan region (McKinney, 2002). In fact, urban areas are the foci for many introduced plant and animal species (Kowarik and von der Lippe, 2007), some of which are invasive and can have large effects on soil processes (Ehrenfeld, 2003; Lilleskov et al., 2010). Additionally, urban land uses can act as "reservoirs" of invasive species populations that are adapted to warmer urban environments, which are then poised to invade peri-urban areas as they become warmer with climate change (e.g., Steinberg et al., 1997).

Direct effects

Land use changes

Converting agriculture, forest, and grasslands to urban and suburban land uses entails a complex array of land and ecosystem alterations. Dense human inhabitation along with urban land-use change necessitates the construction of various built structures such as roads, buildings, and civil infrastructure, as well as the introduction of human activities. Urban development of land typically includes clearing of existing vegetation, grading of soil, importation of fill materials, and the development of structures. The extent and magnitude of these initial disturbances is dependent on topography, infrastructure requirements, and other site limiting factors (Pouyat et al., 2010).

The resultant spatial pattern of disturbance and management practices, or direct effects, are largely the result of "parcelization," or the subdivision of land by ownership, as the landscape undergoes human settlement (Pouyat et al., 2007). This parcelization creates distinct parcels with characteristic disturbance and management regimes that over time affect soils and plant cover. The net result is a mosaic of soil patches and associated plant cover with each patch varying in size and configuration dependent on human population density, development pattern, and transportation networks, among other factors (Pouyat et al., 2010). The overall impact of direct effects occurring at various intensities within the soil mosaic results in a wide range of soil characteristics and plant cover, the latter being expected to respond more rapidly to climate change (Fig. 10.2).

Physical changes

While few data address the effects of disturbance and management associated with urban development on various soil characteristics, especially soil organic C, a plethora of data from studies conducted on cultivated soils sheds light on the potential for soil physical changes occurring in urban landscapes. As an example, native ecosystems converted by cultivation often have reduced levels of soil organic C due to physical disturbance, removal of organic material by harvest, and changes in litter quality that increase decomposition rates (Mann, 1986). Thus, the primary effect of a physical disturbance is a reduction of physical protection of soil organic C by soil aggregates, resulting in C losses. Additionally, highly disturbed urban soils often have massive or platy structure and high bulk densities that can limit aeration and water availability (Craul, 1992). Hence, in urban areas, the physical impacts of a host of direct effects on soils is not expected to be significantly altered by climate change (Fig. 10.2).

Resource changes

In addition to a loss in structure, soils in urban landscapes are often fertilized and watered, especially those soils associated with residential lawns or planting beds. In agricultural and grassland ecosystems, fertilization generally increases net primary production and soil organic C content (Conant et al., 2001). In urban turfgrass systems, the response of soil organic C to fertilizer inputs has been mixed (Pouyat et al., 2007); however, irrigation of turfgrass increases soil organic C in comparison to native grasslands, especially in drier regions (Golubiewski, 2006).

While highly impacted soils typically exhibit low organic matter content and high bulk densities, these characteristics tend to be ameliorated for plant growth in highly maintained soils. Therefore, it is expected that as the climate changes, human efforts to reduce plant-water stress by adjusting irrigation rates will limit a soil's response when it is managed, especially for those soil characteristics related to plant detrital inputs (Fig. 10.2; Pouyat et al., 2007).

Urban environments as analogs to climate change

The interactive and cumulative effects of multiple atmospheric and climate factors on soil and plant health is a major uncertainty in global change research due to the difficulty of implementing controlled, factorial experiments in ecosystems of relatively high complexity (Norby and Luo, 2004). As already mentioned, urban environments exhibit to a large degree many global climate factors undergoing change due to human activities. Because of this similarity, it has been proposed that urban areas can serve as analogs of future climate conditions, particularly in cases where remnant ecosystems (e.g., a forest patch embedded in an urban landscape) are comparable to similar ecosystems situated along "urban-rural gradients" (McDonnell and Pickett, 1990; Carreiro and Tripler, 2005). These "whole ecosystem" comparisons offer the advantage of assessing multiple factors, their interactions, and potential for feedbacks that typically occur in complex ecological systems (Pouyat et al., 2009).

Urban-rural environmental comparisons

Studies employing urban-rural gradients have shown that soils of remnant forests are altered by environmental changes occurring along the gradient. For instance, forest soils within or near urban areas often receive high amounts of heavy metals, organic compounds, and acidic compounds in atmospheric deposition. Lovett et al. (2000) quantified throughfall deposition of N in oak stands in the New York City, USA, metropolitan area and found that urban remnant forests received up to a two-fold greater N flux than rural forests with a significant deposition drop occurring 45 km from the urban core. Similar results were found for Louisville, USA; the San Bernardino Mountains in the Los Angeles metropolitan area, USA; Oulu, Finland; Kaunas, Lithuania; Boston, USA; and the Pearl River Delta, South China, where N deposition rates, and in some cases S and base cation deposition rates, into urban forest patches were higher than in rural forest patches (Bytnerowicz et al., 1999; Juknys et al., 2007; Carreiro et al., 2009; Rao et al., 2014; Huang et al., 2015).

Depositional patterns along urban-rural gradients have also been inferred from accumulations of various pollutants in forest soils. Pouyat et al. (2008) found up to a two to three-fold increase in concentration of lead (Pb), copper (Cu), and nickel (Ni) in urban compared to in rural forest stands in the New York City and Baltimore, USA metropolitan areas. A similar pattern, but with greater differences, was found by Inman and Parker (1978) in the Chicago, USA, metropolitan area where levels of heavy metals were more than five times higher in urban than in rural forest stands. Other urbanization gradient studies have shown a similar pattern (Watmough et al., 1998; Sawicka-Kapusta et al., 2003), although smaller cities, or cities having more condensed development patterns, exhibited less of a difference between urban and rural remnant forests (e.g., Pavao-Zuckerman and Coleman, 2005; Pouyat et al., 2008; Carreiro et al., 2009). Besides accumulations of heavy metals, Wong et al. (2004) found a steep gradient of Polycyclic Aromatic Hydrocarbons (PAH) concentrations in forest soils in the Toronto, Canada metropolitan area, with concentrations decreasing from the urban center to the surrounding rural area. Similarly, Jensen et al. (2007) and Zhang (2006) found higher concentrations of PAHs in surface soils of Oslo, Norway, and Hong Kong, China, respectively, than in surrounding rural areas.

Only a few comparisons have been made for the temperature of forest soils along urban-rural gradients. In the New York City metropolitan area, surface temperatures (2 cm depth) differed by as much as 3 °C between urban and rural forest patches (Pouyat et al., 2003). Comparing this difference with those of a remnant forest patch and nearby highly disturbed soil (see Mount et al., 1999), the difference between a disturbed urban soil and a rural forest soil can be as high as 6 °C in the New York City metropolitan region (Pouyat et al., 2003). A similar pattern, but with smaller differences was found in soil temperature comparisons made in the Baltimore metropolitan area. Savva et al. (2010) measured soil temperatures for turfgrass and forest remnant soils along an urban-rural gradient and found that average annual soil temperature was higher in urban than rural sites under both turfgrass (15.0 vs. 13.5 °C) and forest (12.6 vs. 12.2 °C) cover. These differences were greater during the growing season with the difference between urban and rural forest soils at approximately 1 °C in the spring and fall seasons.

C and N cycling—examples of whole ecosystem responses

How climate and environmental change will affect the cycling of C and N in forest ecosystems over the long term is an important question in soil conservation and management (Muñoz and Zornoza, 2018). In urban ecosystems, the cycling of C and N is expected to be altered in the next decades by multiple stressors from both urban influences (e.g., pollution stress) and changes in global climate (e.g., drought stress). Indeed, each of these elemental cycles exhibit important feedback mechanisms through their emissions as greenhouse gases, e.g., CO_2, CH_4, and nitrous oxide (N_2O).

As previously mentioned, one approach to investigating the effects of multiple stress factors resulting from climate change is to compare forest stands chronically exposed to urban environmental factors with similar forests more remote from urban areas, which is reflected in the urban-rural gradient studies previously cited. A conceptual model of C and N pools and fluxes in forest soils (Fig. 10.3) was

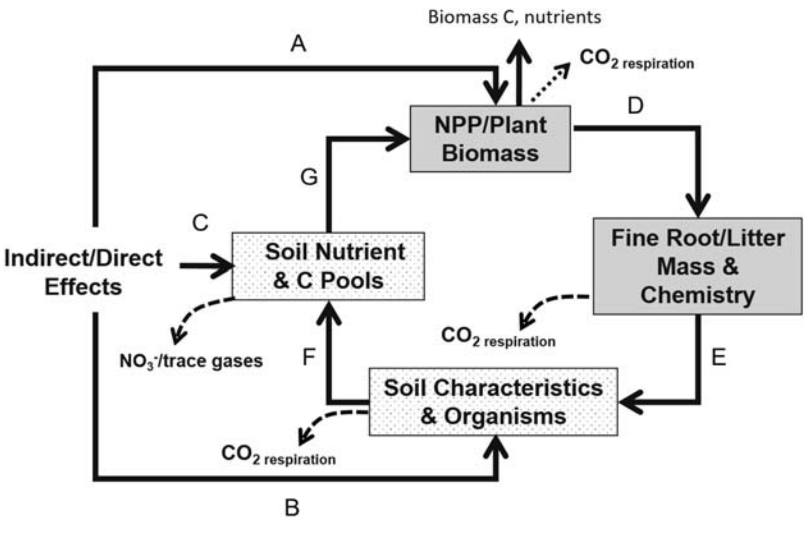

FIG. 10.3

Conceptual diagram of the effects of urban land use on carbon (C) and nitrogen (N) cycling. *Dashed lines* represent losses from the ecological system. *Solid lines* represent inter-connective relationships, such as stresses and enrichment between the nutrient pools, flora, and fauna (*boxes*), and are explained in the text. Stippled *shaded* and *gray boxes* represent potential aboveground and belowground components that can be effected by urban factors, respectively.

Modified from Yesilonis and Pouyat (2012).

developed to explain observations along urban-rural gradients in the New York, NY, Baltimore, MD and Louisville, KY metropolitan areas (Carreiro et al., 2009; Yesilonis and Pouyat, 2012). In the model, both the C and N cycles are coupled through the chemical composition and quantity of leaf litter. As leaf litter increases in quality (e.g., increased concentration of N in ratio to C or lignin), the higher the decomposition of organic matter and thus N availability to plants and soil respiration rates (E and F in Fig. 10.3). In turn, the higher N availability to plants increases their net primary productivity (NPP in Fig. 10.3) and depending on their N-use efficiency may increase or decrease concentrations of leaf N. However, as more N becomes available to soil microbes the demand for N ultimately is exceeded, resulting in excess losses of N from the system (e.g., leaching of nitrate or higher N_2O fluxes), which is referred to as a state of N saturation (Aber et al., 1989). Additionally, soil microbes are limited by daily and seasonal changes in soil moisture and temperature, which for urban environments typically results in longer growing seasons and higher maximum temperatures particularly at night (B in Fig. 10.3).

How urban forest remnants respond to both urban and climate change factors will depend on how the multiple environmental factors will interact in their effect on plant and decomposer responses (Yesilonis and Pouyat, 2012). Several response scenarios are possible given the relationship of the model components in Fig. 10.3. For instance, decomposition rates can be stimulated by both urban and climate change factors (e.g., higher N deposition and soil temperatures), thus resulting in higher N availability in the soil, which will increase NPP and N concentrations in litter, as long as N is limiting growth. By contrast, higher O_3 (urban effect) and lowered soil moisture (climate effect) should depress decomposition rates and thus reduce N availability in the soil, which in turn lowers concentrations of N in litter (D in Fig. 10.3). Other scenarios include lowered NPP and soil organism activity due to pollution stress (A and B in Fig. 10.3), which should slow decay and N transformation rates (F in Fig. 10.3). Additionally, the potential for feedbacks can be examined such as increases or decreases in fine-root production in response to higher inputs of N (Nadelhoffer, 2000), N inhibition of the production of lignin digesting enzymes (Carreiro et al., 2000), or increases in C or lignin to N ratios in leaf litter produced under higher atmospheric concentrations of CO_2 (Hyvönen et al., 2007) (D and E in Fig. 10.3). Finally, the conceptual model allows for the potential influence of climate and urban environmental changes on the successful introductions of various soil invertebrates or invasive plant species (A and B in Fig. 10.3).

Implications to plant health

Urban ecosystems are ideal for studying the combined impact of multiple global change factors on plant growth and reproduction, yet few studies have utilized urban systems to understand plant response to higher temperatures, atmospheric CO_2 concentrations, and N deposition (Calfapietra et al., 2015). Previous research on non-woody plants suggests urban forests may respond to warmer temperatures and greater atmospheric CO_2 concentrations with enhanced growth (Ziska et al., 2003, 2004). However, Gregg et al. (2003) showed higher urban tree growth relative to rural trees was due to a dampening effect of atmospheric O_3 on the rural tree growth rather than urban trees responding positively to warmer temperatures, greater CO_2 and elevated N deposition, which is consistent with CO_2 enrichment experiments (e.g., Wustman et al., 2001). The greatest effect of warmer urban temperatures on plant growth is expected to occur through a longer growing season since the greatest UHI effect occurs at night when plant stomata are closed (Neil and Wu, 2006; Yesilonis and Pouyat, 2012). Increasing temperatures due to global change should continue to alter urban plant phenology, including the timing of leaf out and the duration of full canopy conditions. However, the maximum growing season length tree species will encounter before species shifts occur is not well understood since changes in leaf duration related to increased temperature varies by species (Xu et al., 2014). The response of urban forest remnants to elevated global temperatures will be influenced by other global change factors, such as higher atmospheric CO_2 and N deposition, which already influence forests within urban environments (A in Fig. 10.3).

Forests are an important global sink for C and N (Galloway et al., 2008; Pan et al., 2011). Greater atmospheric CO_2 and N deposition could provide a fertilizer effect on plant growth in urban forest remnants, yet plant growth can be constrained by urban conditions that decrease stomatal opening and N uptake, such as soil moisture constraints. In fact, urban trees had significantly higher N resorption efficiency and proficiency than rural trees (Trammell and Carreiro, 2015), which indicates reduced N availability in these urban remnants, in line with the slower N mineralization rates observed in the urban compared to rural soils (Carreiro et al., 2009; Trammell et al., 2017). Research on forest ecosystems suggest chronic N deposition enhances tree growth and organic matter inputs to soil, hence soil C storage (de Graaff et al., 2006; Frey et al., 2014; Pinder et al., 2012). While these findings help formulate hypotheses about urban tree foliar chemistry and productivity and subsequent soil processes in response to climate change (Fig. 10.3), there still remains a limited understanding of how coupled tree-soil C and N feedbacks respond to multiple urban environmental conditions.

Implications to soil health

How global climate and regional urban environmental factors interact to affect soil organisms or soil biological processes is unclear, but results thus far from observations across urban-rural gradients suggest that the effects depend on the importance of individual factors, which in turn are dependent on the development pattern of the city, occurrence of industrial land uses, and other socio-economic factors (Pouyat et al., 2007; Carreiro et al., 2009). For example, Inman and Parker (1978) found slower leaf litter decomposition rates in urban stands that were highly contaminated with Cu (76 mg kg^{-1}) and Pb (400 mg kg^{-1}) compared to unpolluted rural stands, suggesting a negative pollution effect in the Chicago, USA metropolitan area with a history of heavy industrial activity. Likewise, for a heavily industrialized city in northern Finland, which had excessively high rates of S deposition, several soil biological measurements such as microbial biomass and decay rates were negatively affected in forests near the city (Ohtonen, 1994). By contrast, Pouyat et al. (1997), Pouyat and Turechek (2001) and Zhu and Careirro (2004) in the New York City, USA metropolitan area found higher decomposition and N transformation rates in urban compared to rural oak forest stands, which were attributed to the presence of invasive earthworm species, higher N availability and longer growing seasons in spite of moderate contamination of trace metals in those soils. Similarly, decay rates, soil respiration, and soil N-transformation increased in forest stands near or within major metropolitan areas of the USA in southern California (Fenn and Dunn, 1989), Ohio (Kuperman, 1999), southeastern New York (McDonnell et al., 1997; Carreiro et al., 2009), and Maryland (Groffman et al., 2006; Szlavecz et al., 2006). Therefore, it appears that when heavy metal contamination of soil is moderate to low relative to other atmospherically deposited elements such as N, biological activity may actually be stimulated, particularly where the UHI effect extends the growing season. These higher rates of biological activity appear to translate to higher trace gas fluxes from urban forest remnants (Groffman et al., 2006; Groffman and Pouyat, 2009), which is similar with fluxes measured from managed urban soils (e.g., Kaye et al., 2004).

Results from experimental soil warming studies suggest that soil microbial activity and soil ecosystem processes in urban forest remnants may be enhanced due to the UHI effect (Craine et al., 2010; Butler et al., 2012). Similarly, in a soil transplant study associated with an urban-rural gradient, Pouyat and Turechek (2001) found that soil temperature differences between urban and rural forest soils accounted for up to 20% of the enhanced nitrification rates measured in the urban stands. Even with these results, the net effect of soil warming on microbial processes and N and C cycles (e.g., net soil CO_2 flux) will depend on other urban environmental factors. As previously mentioned, another impact of increases in ambient temperature, especially during the day, is the formation of atmospheric O_3, which has been shown to reduce NPP and impact leaf litter quality (Findlay et al., 1996; Gregg et al., 2003).

Furthermore, invasive species can play a disproportionate role in controlling C and N cycles in terrestrial ecosystems (Ehrenfeld, 2003; Bohlen et al., 2004). Therefore, the relationship between invasive species abundances and urban land uses has important implications for soil-mediated ecosystem processes (Pouyat et al., 2007). For example, in the northeastern and mid-Atlantic United States where native earthworm species are rare or absent, urban areas are important foci of invasive earthworm introductions, especially Asian species from the genus *Amynthas*, which are expanding their range to outlying forest areas (Steinberg et al., 1997; Groffman and Bohlen, 1999; Szlavecz et al., 2006). Invasions by earthworms into forests have resulted in highly altered C and N cycling processes (Bohlen et al., 2004; Hale et al., 2005; Carreiro et al., 2009; Sackett et al., 2013). Likewise, plant species invasions can impact C and N cycles (Liao et al., 2008), which in some cases can facilitate the colonization of additional invasive species, such as earthworms, further exacerbating the turnover of N in the soil (Pavao-Zuckerman, 2008). Examples of plant invasions in urban metropolitan areas that have altered C and N cycles include species of shrubs *Berberis thunbergii* and *Lonicera maackii*, the tree *Rhamnus cathartica*, and the grass *Microstegium vimineum* (Ehrenfeld et al., 2001; Heneghan et al., 2002; Trammell et al., 2012).

Urban soils: the brown infrastructure of cities and towns

Natural landscapes embedded within urban and suburban development are necessary for providing ecosystem services to humans and other biota. While green infrastructure receives considerable attention for ecosystem service provision in cities and towns (e.g., Pataki et al., 2011; Lovell and Taylor, 2013), the underlying soil structure and function are vital for the functioning and delivery of these benefits along the USC (Fig. 10.1). The importance of soils in mitigating environmental change by providing ecosystem services has recently been gaining ground in urban areas (e.g., Anne et al., 2018). The term, "brown infrastructure" has been proposed by Pouyat et al. (2010) to emphasize the equivalent importance of soils to the overlying greenspace in urban landscapes (i.e., green infrastructure). The ability of greenspace in cities to provide ecosystem services, such as water infiltration, C storage, and recreation, is dependent on underlying soil conditions and health (Pavao-Zuckerman, 2012). Similarly, soils are vital as the foundation for gray infrastructure within cities, i.e., roads and buildings. Therefore, soils deserve recognition for the essential services provided within developed landscapes.

Ecosystem services

Maintaining biodiversity

As previously mentioned, urban soils experience altered temperatures and increased chemical deposition (Fig. 10.2) creating soil conditions (e.g., soil drought, heavy metals) that could support species tolerant of dry and/or contaminated sites, such as non-native invasive tree species. Additionally, land use changes create a mosaic of urban forest remnants differing in age (e.g., time since agriculture abandonment), size (e.g., park vs. roadside), and configuration (e.g., linear vs. square). Despite these factors creating a mosaic of soil conditions, urban forest remnants maintain a diverse, native tree canopy (Trammell and Carreiro, 2011, Trammell et al., 2019), and, as such, may provide some resiliency to urban environmental and chemical changes. The diversity-stability relationship suggests that diverse communities can protect against fluctuations in environmental conditions as well as being more resistant to potential pest outbreaks. Therefore, the brown infrastructure supporting native tree biodiversity in urban forests is vital in cities and towns.

Maintaining biogeochemical cycles

Due to their proximity to anthropogenic activities that increase elemental cycling, urban forests function with greater C, N, and other nutrient cycles (Kay et al., 2006). Greater atmospheric concentrations of CO_2 are thought to enhance forest productivity (Pan et al., 2011); however, this potential is dependent on whether other nutrients necessary for forest production, such as N, are in sufficient supply for optimal plant growth and for maintaining soil processes. A meta-analysis of CO_2 enrichment studies found that elevated CO_2 enhanced microbial activity only occurred with high N availability (de Graaff et al., 2006) suggesting soil microbial activity can persist within altered chemical environments of urban forest remnants (Fig. 10.3). Without direct human manipulation and/or management, urban remnants embedded within nutrient rich environments can maintain natural biogeochemical cycles. Thus, soil processes, such as C and N soil mineralization; that are important for regulating elemental cycles are maintained within these highly developed ecosystems.

Pollution storage

As shown with observations along urban-rural gradients, urban forest remnants characteristically receive greater pollutant deposition than nearby rural forests (Lovett et al., 2000; Carreiro et al., 2009). Many urban ecosystems have greater heavy metal concentrations from current (e.g., traffic) and legacy (e.g., industry) activities that accumulate in soils (Yesilonis et al., 2008). Urban forest soils can act as an important sink for the transport and storage of heavy metal pollutants since concentrations in urban forests rarely exceed values that dampen microbial functioning (Bååth, 1989). However, land use change and/or development of forests within cities or in peri-urban areas can pose a significant health risk with the redistribution of heavy metals following soil disturbance, especially in urban forest soils that may contain legacy lead from the leaded-gasoline era (Schwarz et al., 2016). Therefore, conservation of existing urban forest remnants may not only be vital for maintaining current ecosystems services associated with heavy metal pollutant capture, but may also be important for long-term storage of legacy pollutant inputs.

As discussed earlier, several studies have shown elevated N deposition occurring in urban forest remnants. Forest ecosystems are an important sink for reactive N deposition (Galloway et al., 2008) unless a threshold of N saturation is reached where the forest can become an N source (Aber et al., 1989). Multiple aspects of urbanization, such as invasive species spread, climate variation, and altered nutrient cycling, interact and feedback to alter forest structure and function, ultimately determining the ability of forests to act as an N sink (Fig. 10.3). Previous research found increased soil N cycling rates in urban compared to rural forest remnants (e.g., Pouyat et al., 1997; Zhu and Careirro, 2004; Pavao-Zuckerman and Coleman, 2005) suggesting urban and suburban remnants have a greater potential for N losses than rural remnants (Pouyat and Turechek, 2001). However, soil N cycling slowed in the urban forest remnants of one mid-size city suggesting urban remnants may be an important N sink in this urban system (Trammell et al., 2017). The potential for forests to act as an N sink in urban environments does depend on the soil conditions (e.g., non-native invasive species, soil moisture) that control N cycling rates and losses.

Soils mitigate global change

Global environmental change drivers, such as increased temperatures, altered precipitation patterns, and invasive species spread, are currently affecting soils in urban forest remnants (Fig. 10.2). Soils have the potential to buffer some consequences of global change by maintaining biogeochemical cycles that provide water and nutrients for plant health and growth. The ability of urban soils to provide resiliency for future global changes in climate will depend on the current conditions in forests exposed to current urban warming and soil moisture constraints.

Water storage

Forests are essential for global and regional water cycles due to their high capacity to store and redistribute water. Evapotranspiration by forest trees is an important cooling mechanism that can alleviate warming temperatures from urbanization and global climate change (e.g., Bowler et al., 2010). This essential ecosystem service is, however, dependent on tree condition and soil moisture status. Since a higher proportion of tree canopy cover within urban environments has the potential to provide greater cooling benefits in cities (Forman, 2014), it is reasonable to expect larger forest remnants would buffer against moisture constraints due to the urban heat island. Research is needed to understand how size, shape, amount, and location of urban forest remnants can provide maximum water storage and how higher urban temperatures may alter soil moisture and, ultimately, forest productivity.

Urban forest functioning is not only dependent on current anthropogenic activities but also on past land use and cover. Urban forest remnants (Zipperer, 2002; Trammell et al., in review) and other urban land uses (e.g., residential yards; Raciti et al., 2011) experience a range of land use history across developed landscapes. Within the eastern portion of the United States, agriculture is a common previous land use for many small forests patches located within cities and towns (Vellend, 2003). While forests demonstrate recovery of tree diversity and/or soil C and N cycling within 100 years of agriculture abandonment, the lack of long-term tree uprooting, the most obvious natural cause for micro-scale topographic relief in forests, remains absent (Flinn and Marks, 2007). This lack of microrelief could have substantial effects on runoff by reducing water infiltration to deeper soils (Valtera and Schaetzl, 2017). Altered soil structure due to tillage practices could also provide long-term legacies on soil structure and pore space, which are essential for water storage potential.

Carbon storage

The importance of soils in mitigating climate change via C storage and sequestration has received a great deal of attention, and the potential of urban soil to mitigate global change via C storage is gaining ground (e.g., Pouyat et al., 2006; Ziter and Turner, 2018). The C storage and sequestration potential of urban soils varies across land use and land cover in cities as numerous direct and indirect anthropogenic influences can increase or decrease urban soil C (Fig. 10.3; Pouyat et al., 2006; Trammell et al., 2018). Indeed, research on practices to restore C in what were highly disturbed soils have been encouraging (e.g., Chen et al., 2014). Even without human intervention, the UHI can increase the green phase of urban plants and potentially enhance plant productivity (e.g., Neil and Wu, 2006; Ziska et al., 2004) thereby increasing organic matter and root exudate inputs to soil (Fig. 10.3). On the other hand, greater urban temperatures can also enhance decomposition rates, thus trace gas emissions, offsetting the effect of greater plant productivity on soil C storage. Biotic factors, such as species shifts, can also affect the rates of organic matter quality and inputs to soil and soil microbial activity. For example, non-native invasive plants dampen soil C storage in forests adjacent to urban interstates (Trammell and Carreiro, 2012). Whether urban soils store and sequester significant amounts of C will depend on many factors that impact plant productivity relative to soil microbial activity (Trammell et al., 2018).

Soil organism resiliency

During the last century, the rapid growth of urban areas into surrounding regions has created vast areas of suburban and exurban lands. Through this urban expansion, temperature changes due to local and regional human activities may progress at a rate that could provide soil organisms adequate time to acclimate to altered temperature regimes. As an example, in an experimental study on urban soils from temperate deciduous forests, warming (+2 °C) did not alter soil N mineralization rates (Trammell et al., 2017) suggesting that with all other factors being equal, soil microbes may be resilient to small changes in temperature in urban forest remnants. While more research is needed to address urban soil function with respect to predicted changes in climate, microbial functioning in urban forest soils suggests these organisms may be resilient to small changes in temperature and precipitation in the future.

Summary and conclusions

Urban soils are becoming increasingly important as more than half of the world's human population now live in urban areas, a percentage that is expected to rise in the foreseeable future. Urban environmental factors, both indirect and direct, can have significant and quantifiable effects on soil characteristics in and near urbanized areas. These impacts range from highly disturbed and managed soils to relatively unmodified soils, forming a continuum of soil conditions – the Urban Soil Continuum. Direct effects include those generally associated with soil modifications occurring on the more highly disturbed end of the continuum, while indirect effects involve changes in the abiotic and biotic environment that influence mostly undisturbed soils associated with remnant forests or grasslands. Additionally, many urban environmental factors, such as the urban heat island and elevated atmospheric concentrations of CO_2, are similar to factors expected to occur with global climate change. For this reason, urban areas can serve as analogs of future climate conditions. Global climate change is expected to have the largest impact on remnant forest or grassland soils, while the most disturbed and highly designed and maintained urban soils may have the greatest potential to mitigate climate change factors. Soils are the brown infrastructure providing ecosystem services across urban land uses and their role in the mitigation of climate change is vital for the future and resiliency of densely populated regions of the world.

References

Aber, J.D., Nadelhoffer, K.J., Steudler, P., Melillo, J.M., 1989. Nitrogen saturation in northern forest ecosystems. BioScience 39 (6), 378−386.

Ahiablame, L.M., Engel, B.A., Chaubey, I., 2012. Effectiveness of low impact development practices: literature review and suggestions for future research. Water, Air, and Soil Pollution 223, 4253−4273.

Altshullera, A.P., Ortmana, G.C., Saltzmana, B.E., Neliganb, R.E., 1996. Continuous monitoring of methane and other hydrocarbons in urban atmospheres. Journal of the Air Pollution Control Association 16, 87−91.

Anne, B., Geoffroy, S., Cherel, J., Warot, G., Marie, S., Noël, C.J., Louis, M.J., Christophe, S., 2018. Towards an operational methodology to optimize ecosystem services provided by urban soils. Landscape and Urban Planning 176, 1−9.

Bååth, E., 1989. Effects of heavy metals in soil on microbial processes and populations (a review). Water, Air, and Soil Pollution 47, 335−379.

Bockheim, J.G., 1974. Nature and properties of highly-disturbed urban soils. In: Proceedings of Division S-5, Soil Genesis, Morphology and Classification. Annual Meeting of the Soil Science Society of America, Philadelphia, Pennsylvania (Chicago, IL).

Bohlen, P.J., Groffman, P.M., Fahey, T.J., Fisk, M.C., Suarez, E., Pelletier, D.M., Fahey, R.T., 2004. Ecosystem consequences of exotic earthworm invasion of north temperate forests. Ecosystems 7, 1−12.

Bowler, D.E., Buyung-Ali, L., Knight, T.M., Pullin, A.S., 2010. Urban greening to cool towns and cities: a systematic review of the empirical evidence. Landscape and Urban Planning 97, 147−155.

Butler, S.M., Melillo, J.M., Johnson, J.E., Mohan, J., Steudler, P.A., Lux, H., Burrows, E., Smith, R.M., Vario, C.L., Scott, L., Hill, T.D., Aponte, N., Bowles, F., 2012. Soil warming alters nitrogen cycling in a New England forest: implications for ecosystem function and structure. Oecologia 168, 819−828.

Bytnerowicz, A., Fenn, M.E., Miller, P.R., Arbaugh, M.J., 1999. Wet and dry pollutant deposition to the mixed conifer forest. In: Miller, P.R., Mcbride, J.R. (Eds.), Oxidant Air Pollution Impacts in the Montane Forests of Southern California: A Case Study of the San Bernardino Mountains. Springer, New York.

Calfapietra, C., Peñuelas, J., Niinemets, U., 2015. Urban plant physiology: adaptation-mitigation strategies under permanent stress. Trends in Plant Science 20, 72−75.

Capra, C.F., Ganga, A., Grilli, E., Vacca, S., Buondonno, A., 2015. A review on anthropogenic soils from a worldwide perspective. Journal of Soils and Sediments 15, 1602−1618.

Carreiro, M.M., Tripler, C.E., 2005. Forest remnants along urban-rural gradients: examining their potential for global change research. Ecosystems 8, 568−582.

Carreiro, M.M., Sinsabaugh, R.L., Repert, D.A., Parkhurst, D.F., 2000. Microbial enzyme shifts explain litter decay responses to simulated nitrogen deposition. Ecology 81, 2359−2365.

Carreiro, M.M., Pouyat, R.V., Tripler, C.E., Zhu, W., 2009. Carbon and nitrogen cycling in soils of remnant forests along urban-rural gradients: case studies in the New York metropolitan area and Louisville, Kentucky. In:

McDonnell, M.J., Hahs, A.K., Breuste, J.H. (Eds.), Ecology of Cities and Towns: A Comparative Approach. Cambridge University Press, NY, pp. 308−328.

Caspersen, J.P., Pacala, S.W., Jenkins, J.C., Hurtt, G.C., Moorcroft, P.R., Birdsey, R.A., 2000. Contributions of land-use history to carbon accumulation in U.S. forests. Science 290, 1148−1151.

Chen, Y., Day, S.D., Wick, A.F., McGuire, K.J., 2014. Influence of urban land development and subsequent soil rehabilitation on soil aggregates, carbon, and hydraulic conductivity. The Science of the Total Environment 494, 329−336.

Chen, Y., Wang, X., Jiang, B., Yang, N., Li, L., 2016. Pavement induced soil warming accelerates leaf budburst of ash trees. Urban Forestry and Urban Greening 16, 36−42.

Conant, R.T., Paustian, K., Elliott, E.T., 2001. Grassland management and conversion into grassland: effects on soil carbon. Ecological Applications 11, 343−355.

Craine, J.M., Fierer, N., McLauchlan, K.K., 2010. Widespread coupling between the rate and temperature sensitivity of organic matter decay. Nature Geoscience 3, 854−857.

Craul, P.J., 1992. Urban Soil in Landscape Design. John Wiley & Sons, Inc, New York.

Craul, P.J., Klein, C.J., 1980. Characterization of streetside soils of Syracuse. METRIA, New York, pp. 88−101, 3.

Datko-Williams, L., Wilkie, A., Richmond-Bryant, J., 2014. Analysis of U.S. soil lead (Pb) studies from 1970 to 2012. The Science of the Total Environment 468−469, 854−863.

de Graaff, M., Van Groenigen, K.-J., Six, J., Hungate, B., Van Kessel, C., 2006. Interactions between plant growth and soil nutrient cycling under elevated CO_2: a meta-analysis. Global Change Biology 12, 2077−2091.

Decina, S.M., Templer, P.H., Hutyra, L.R., Gately, C.K., Rao, P., 2017. Variability, drivers, and effects of atmospheric nitrogen inputs across an urban area: emerging patterns among human activities, the atmosphere, and soils. The Science of the Total Environment 609, 1524−1534.

Edmondson, J.L., Davies, C.Z., Gaston, K.J., Leake, J.R., 2014. Urban cultivation in allotments maintains soil qualities adversely affected by conventional agriculture. Journal of Applied Ecology 51, 880−889.

Effland, W.R., Pouyat, R.V., 1997. The genesis, classification, and mapping of soils in urban areas. Urban Ecosystems 1, 217−228.

Ehrenfeld, J.G., 2003. Effects of exotic plant invasions on soil nutrient cycling processes. Ecosystems 6, 503−523.

Ehrenfeld, J.G., Kourtev, P., Huang, W., 2001. Changes in soil functions following invasions of exotic understory plants in deciduous forests. Ecological Applications 11, 1278−1300.

Emberson, L.D., Kitwiroon, N., Beevers, S., Büker, P., Cinderby, S., 2013. Scorched Earth: how will changes in the strength of the vegetation sink to ozone deposition affect human health and ecosystems? Atmospheric Chemistry and Physics 13, 6741−6755.

Evans, C.V., Fanning, D.S., Short, J.R., 2000. Human-influenced soils. Agronomy Monograph 39, 33−67.

Fenn, M.E., Dunn, P.H., 1989. Litter decomposition across an air-pollution gradient in the san Bernardino Mountains. Soil Science Society of America Journal 53, 1560−1567.

Findlay, S., Carreiro, M., Krischik, V., Jones, C., 1996. Effects of damage to living plants on leaf litter quality. Ecological Applications 6, 269−275.

Flinn, K.M., Marks, P.L., 2007. Agricultural legacies in forest environments: tree communities, soil properties, and light availability. Ecological Applications 17 (2), 452−463.

Forman, R.T.T., 2014. Urban Ecology: Science of Cities. Cambridge University Press.

Frey, S.D., Ollinger, S., Nadelhoffer, K., Bowden, R., Brzostek, E., Burton, A., Caldwell, B.A., Crow, S., Galloway, J.N., Townsend, A.R., Erisman, J.W., Bekunda, M., Cai, Z., Freney, J.R., Martinelli, L.A., Gill, M., 2014. Heavy metal stress in plants: a review. International Journal of Advanced Research 2, 1043−1055.

Galloway, J.N., Townsend, A.R., Erisman, J.W., Bekunda, M., Cai, Z., Freney, J.R., Martinelli, L.A., Seitzinger, S.P., Sutton, M.A., 2008. Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. Science 889−892.

Gill, M., 2014. Heavy metal stress in plants: a review. International Journal of Advanced Research 2, 1043–1055.
Golubiewski, N.E., 2006. Urbanization increases grassland carbon pools: effects of landscaping in Colorado's front range. Ecological Applications 16, 555–571.
Grabosky, J., Bassuk, N., Marranca, B.Z., 2002. Preliminary findings from measuring street tree shoot growth in two skeletal soil installations compared to tree lawn plantings. Journal of Arboriculture 28, 106–108.
Graves, W.R., 1994. Urban soil temperatures and their potential impact on tree growth. Journal of Arboriculture 20, 24–27.
Gregg, J.W., Jones, C.G., Dawson, T.E., 2003. Urbanization effects on tree growth in the vicinity of New York City. Nature 424, 183–187.
Groffman, P.M., Bohlen, P.J., 1999. Soil and sediment biodiversity cross-system comparisons and large-scale effects. BioScience 49, 139–148.
Groffman, P.M., Pouyat, R.V., 2009. Methane uptake in urban forests and grasslands. Environmental Science and Technology 43, 5229–5235.
Groffman, P.M., Pouyat, R.V., Cadenasso, M.L., Zipperer, W.C., Szlavecz, K., Yesilonis, I.D., Band, L.E., Brush, G.S., 2006. Land use context and natural soil controls on plant community composition and soil nitrogen and carbon dynamics in urban and rural forests. Forest Ecology and Management 236, 177–192.
Hale, C.M., Frelich, L.E., Reich, P.B., Pastor, J., 2005. Effects of European earthworm invasion on soil characteristics in northern hardwood forests of Minnesota, USA. Ecosystems 8, 911–927.
Hall, S.J., Learned, J., Ruddell, B., Larson, K.L., Cavender-Bares, J., Bettez, N., Groffman, P.M., Grove, J.M., Heffernan, J.B., Hobbie, S.E., Morse, J.L., Neill, C., Nelson, K.C., O'Neil-Dunne, J.P.M., Ogden, L., Pataki, D.E., Pearse, W.D., Polsky, C., Roy Chowdhury, R., Steele, M.K., Trammell, T.L.E., 2015. Convergence of microclimate in residential landscapes across diverse cities in the United States. Landscape Ecology 31, 101–117.
Heneghan, L., Clay, C., Brundage, C., 2002. Observations on the initial decomposition rates and faunal colonization of native and exotic plant species in an urban forest fragment. Ecological Restoration 20, 108–111.
Homer, C.G., Dewitz, J.A., Yang, L., Jin, S., Danielson, P., Xian, G., Coulston, J., Herold, N.D., Wickham, J.D., Megown, K., 2015. Completion of the 2011 National Land Cover Database for the conterminous United States-Representing a decade of land cover change information. Photogrammetric Engineering and Remote Sensing 81, 345–354.
Huang, J., Zhang, W., Mo, J., Wang, S., Liu, J., Chen, H., 2015. Urbanization in China drives soil acidification of Pinus massoniana forests. Scientific Reports 5, 13512.
Hyvönen, R., Ågren, G., Linder, S., Persson, T., Cotrufo, M.F., Ekblad, A., Freeman, M., Grelle, A., et al., 2007. The likely impact of elevated (CO_2), nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: a literature review. New Phytologist 173, 463–480.
Imhoff, M.L., Zhang, P., Wolfe, R.E., Bounoua, L., 2010. Remote sensing of the urban heat island effect across biomes in the continental USA. Remote Sensing of Environment 114, 504–513.
Inman, J.C., Parker, G.R., 1978. Decomposition and heavy metal dynamics of forest litter in northwestern Indiana. Environmental Pollution 17, 34–51.
Jensen, H., Reimann, C., Finne, T.E., Ottesen, R.T., Arnoldussen, A., 2007. PAH-concentrations and compositions in the top 2 cm of forest soils along a 120 km long transect through agricultural areas, forests and the city of Oslo, Norway. Environmental Pollution 145, 829–838.
Jochner, S., Menzel, A., 2015. Urban phenological studies – past, present, future. Environmental Pollution 203, 250–261.
Juknys, R., Zaltauskaite, J., Stakenas, V., 2007. Ion fluxes with bulk and throughfall deposition along an urban-suburban-rural gradient. Water, Air, and Soil Pollution 178, 363–372.
Kay, 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.

Kaye, J.P., Burke, I.C., Mosier, A.R., Guerschman, J.P., 2004. Methane and nitrous oxide fluxes from urban soils to the atmosphere. Ecological Applications 14, 975–981.

Koerner, B., Klopatek, J., 2002. Anthropogenic and natural CO_2 emission sources in an arid urban environment. Environmental Pollution 116, S45–S51.

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

Kowarik, I., von der Lippe, M., 2007. Pathways in plant invasions. In: Nentwig, W. (Ed.), Biological Invasions. Ecological Studies, vol. 193. Springer, Berlin, New York, pp. 29–47.

Kuperman, R.G., 1999. Litter decomposition and nutrient dynamics in oak-hickory forests along a historic gradient of nitrogen and sulfur deposition. Soil Biology and Biochemistry 31, 237–244.

Lehmann, A., Stahr, K., 2007. Nature and significance of anthropogenic urban soils. Journal of Soils and Sediments 7, 247–260.

Liao, C., Peng, R., Luo, Y., Zhou, X., Wu, X., Fang, C., Chen, J., Li, B., 2008. Altered ecosystem carbon and nitrogen cycles by plant invasion: a meta-analysis. New Phytologist 177, 706–714.

Lilleskov, E., Callaham Jr., M.A., Pouyat, R.V., Smith, J.E., Castellano, M., González, G., Lodge, D.J., Arango, R., Green, F., 2010. Invasive soil organisms and their effects on belowground processes. Pages 67–84. In: Dix, M.E., Britton, K. (Eds.), A Dynamic Invasive Species Research Vision: Opportunities and Priorities 2009–29. General Technical Report WO-79. USDA Forest Service, Washington, DC.

Lovell, S.T., Taylor, J.R., 2013. Supplying urban ecosystem services through multifunctional green infrastructure in the United States. Landscape Ecology 28, 1447–1463.

Lovett, G.M., Traynor, M.M., Pouyat, R.V., Carreiro, M.M., Zhu, W., Baxter, J.W., 2000. Atmospheric deposition to oak forests along an urban-rural gradient. Environmental Science and Technology 34, 4294–4300.

Mann, L.K., 1986. Changes in soil carbon storage after cultivation. Soil Science 142, 279–288.

McDonnell, M.J., Pickett, S.T.A., 1990. Ecosystem structure and function along urban-rural gradients: an unexploited opportunity for ecology. Ecology 71, 1232–1237.

McDonnell, M.J., Pickett, S.T.A., Groffman, P., Bohlen, P.J., Pouyat, R.V., Zipperer, W.C., Parmelee, R.W., Carreiro, M.M., Medley, K., 1997. Ecosystem processes along an urban-to-rural gradient. Urban Ecosystems 1, 21–36.

McKinney, M.L., 2002. Urbanization, biodiversity, and conservation. BioScience 52, 883–890.

Meinshausen, M., Smith, S.J., Calvin, K., Daniel, J.S., Kainuma, M.L.T., Lamarque, J.-F., Matsumoto, K., Montzka, S.A., Raper, S.C.B., Riahi, K., Thomson, A., Velders, G.J.M., van Vuuren, D.P.P., 2011. The RCP greenhouse gas concentrations and their extensions from 1765 to 2300. Climatic Change 109, 213–241.

Morel, J.L., Burghardt, W., Kim, K.H.J., 2017. The challenges for soils in the urban environment. In: Levin, M.J., Kim, K.H.J., Morel, J.L., Burghardt, W., Charzynski, P., Shaw, R.K. (Eds.), Soils within Cities. Catena Soil Sciences, Schweizerbart, Stuttgart, pp. 1–6.

Mount, H., Hernandez, L., Goddard, T., Indrick, S., 1999. Temperature signatures for anthropogenic soils in New York city. Classification, correlation, and management of anthropogenic soils. In: Proceedings, Nevada and California, September 21 October, 2, 1998, pp. 137–140.

Muñoz, M.A., Zornoza, R., 2018. Soil Management and Climate Change. Academic Press, Cambridge, MA, p. 396.

Nadelhoffer, K.J., 2000. The potential effects of nitrogen deposition on fine-root production in forest ecosystems. New Phytologist 147, 131–139.

National Research Council, 2009. Urban Stormwater Management in the United States. The National Academies Press, Washington, DC. https://doi.org/10.17226/12465.

Neil, K., Wu, J., 2006. Effects of urbanization on plant flowering phenology: a review. Urban Ecosystems 9, 243–257.

Niyogi, D., Pyle, P., Lei, M., Chen, F., Wolfe, B., 2011. Urban modification of thunderstorms: an observational storm climatology and model case study for the Indianapolis urban region. Journal Applied Meteorology and Climatology 50, 1−13.

Norby, R.J., Luo, Y., 2004. Evaluating ecosystem responses to rising CO2 and global warming in a multi-factor world. New Phytologist 162, 281−293.

Ohtonen, R., 1994. Accumulation of organic matter along a pollution gradient: application of Odum's Theory of Ecosystem Energetics. Microbial Ecology 27, 43−55.

Oke, T.R., 1990. The micrometeorology of the urban forest. The Quarterly Journal of the Royal Meteorological Society 324, 335−349.

Pan, Y., Birdsey, R.A., Fang, J., Houghton, R., Kauppi, P., Kurz, W.A., Phillips, O.L., Shvidenko, A., Lewis, S.L., Canadell, J.G., Ciais, P., Jackson, R.B., Pacala, S.W., McGuire, A.D., Piao, S., Rautiainen, A., Sitch, S., Hayes, D., 2011. A large and persistent carbon sink in the world's forests. Science 333, 988−993.

Pataki, D.E., Bowling, D.R., Ehleringer, J.R., 2003. Seasonal cycle of carbon dioxide and its isotopic composition in an urban atmosphere: anthropogenic and biogenic effects. Journal of Geophysical Research − Atmospheres 108 (D23), 4735.

Pataki, D.E., Carreiro, M.M., Cherrier, J., Grulke, N.E., Jennings, V., Pincetl, S., Pouyat, R.V., Whitlow, T.H., Zipperer, W.C., 2011. Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions. Frontiers in Ecology and the Environment 9 (1), 27−36.

Pavao-Zuckerman, M.A., 2008. The nature of urban soils and their role in ecological restoration in cities. Restoration Ecology 16, 642−649.

Pavao-Zuckerman, M., 2012. Urbanization, soils, and ecosystem services. In: Wall, D.H., Bardgett, R.D., Behan-Pelletier, V., Herrick, J.E., Jones, H.P., Ritz, K., Six, J., Strong, D.R., van der Putten, W.H. (Eds.), Soil Ecology and Ecosystem Services. Oxford University Press, Oxford, pp. 270−281.

Pavao-Zuckerman, M.,A., Coleman, D.C., 2005. Decomposition of chestnut oak (*Quercus prinus*) leaves and nitrogen mineralization in an urban environment. Biology and Fertility of Soils 41, 343−349.

Pinder, R.W., Davidson, E.A., Goodale, C.L., Greaver, T.L., Herrick, J.D., Liu, L., 2012. Climate change impacts of US reactive nitrogen. Proceedings of the National Academy of Sciences United States of America 109, 7671−7675.

Pouyat, R.V., Turechek, W.W., 2001. Short- and long-term effects of site factors on net N-mineralization and nitrification rates along an urban-rural gradient. Urban Ecosystems 5 (3), 159−178.

Pouyat, R.V., McDonnell, M.J., Pickett, S.T.A., 1997. Litter decomposition and nitrogen mineralization along an urban-rural land use gradient. Urban Ecosystems 1, 117−131.

Pouyat, R.V., Groffman, P.M., Russell-Anneli, J., Yesilonis, I., 2003. Soil carbon in urban forest ecosystems. In: Lal, R., Kimble, J., Follett, R.F., Birdsey, R. (Eds.), Potential of United States Forest Soils to Sequester Carbon and Mitigate the Greenhouse Effect. CRC Press, NJ, pp. 347−362.

Pouyat, R.V., Yesilonis, I.D., Nowak, D.J., 2006. Carbon storage by urban soils in United States. Journal of Environmental Quality 35, 1566−1575.

Pouyat, R.V., Belt, K., Pataki, D., Groffman, P.M., Hom, J., Band, L., 2007. Urban land-use change effects on biogeochemical cycles. In: Canadell, J.G., Pataki, D.E., Pitelka, L.F. (Eds.), Terrestrial Ecosystems in a Changing World. Global Change, the IGBP Series. Springer, Berlin-Heidelberg-New York, pp. 45−58.

Pouyat, R.V., Yesilonis, I.D., Szlavecz, K., Csuzdi, C., Hornung, E., Korsós, Z., Russell-Anelli, J., Giorgio, V., 2008. Response of forest soil properties to urbanization gradients in three metropolitan areas. Landscape Ecology 23, 1187−1203.

Pouyat, R.V., Carreiro, M.M., Groffman, P.M., Zuckerman, M., 2009. Investigative approaches to urban biogeochemical cycles: New York metropolitan area and Baltimore as case studies. In: McDonnell, M.J., Hahs, A., Breuste, J. (Eds.), Ecology of Cities and Towns: A Comparative Approach. Cambridge University Press, Cambridge, Oxford, pp. 329−354.

Pouyat, R.V., Szlavecz, K., Yesilonis, I.D., Groffman, P.M., Schwarz, K., 2010. Chemical, physical, and biological characteristics of urban soils. In: Aitkenhead-Peterson, J., Volder, A. (Eds.), Urban Ecosystem Ecology (Agronomy Monograph 55). ASA-CSSA-SSSA, Madison, WI, pp. 119−152.

Raciti, S.M., Groffman, P.M., Jenkins, J.C., Pouyat, R.V., Fahey, T.J., Pickett, S.T., Cadenasso, M.L., 2011. Accumulation of carbon and nitrogen in residential soils with different land-use histories. Ecosystems 14 (2), 287−297.

Rao, P., Hutyra, L.R., Raciti, S.M., Templer, P.H., 2014. Atmospheric nitrogen inputs and losses along an urbanization gradient from Boston to Harvard Forest, MA. Biogeochemistry 121, 229−245.

Sackett, T.E., Smith, S.M., Basiliko, N., 2013. Indirect and direct effects of exotic earthworms on soil nutrient and carbon pools in North American temperate forests. Soil Biology and Biochemistry 57, 459−467.

Savva, Y., Szlavecz, K., Pouyat, R.V., Groffman, P.M., Heisler, G., 2010. Land use and vegetation cover effects on soil temperature in an urban ecosystem. Soil Science Society of America Journal 74, 469−480.

Sawicka-Kapusta, K., Zakrzewska, M., Bajorek, K., Gdula-Argasinska, J., 2003. Input of heavy metals to the forest floor as a result of Cracow urban pollution. Environment International 28, 691−698.

Scalenghe, R., Marsan, F.A., 2009. The anthropogenic sealing of soils in urban areas. Landscape and Urban Planning 90, 1−10.

Schwarz, K., Pouyat, R.V., Yesilonis, I.D., 2016. Legacies of lead in charm city's soil: lessons from the Baltimore Ecosystem Study. International Journal of Environmental Research and Public Health 13, 209.

Shaw, R.K., Isleib, J.T., 2017. The case of the New York city soil survey program. In: Levin, M.J., Kim, K.-H.J., Morel, J.L., Burghardt, W., Charzynski, P., Shaw, R.K. (Eds.), IUSS Working Group SUITMA. Soils within Cities—Global Approaches to Their Sustainable Management—Composition, Properties, and Functions of Soils of the Urban Environment. Catena Soil Sciences, Schweizerbart, Stuttgart, pp. 107−113.

Shem, W., Shepherd, M., 2009. On the impact of urbanization on summertime thunderstorms in Atlanta: two numerical model case studies. Atmospheric Research 92, 172−189.

Shepherd, J.M., Burian, S.J., 2003. Detection of urban-induced rainfall anomalies in a major coastal city. Earth Interactions 7, 1−14.

Short, J.R., Fanning, D.S., Foss, J.E., Patterson, J.C., 1986. Soils of the Mall in Washington, DC: II. Genesis, classification and mapping. Soil Science Society of America Journal 50, 705−710.

Song, Y., Liu, H., Wang, X., Zhang, N., Sun, J., 2016. Numerical simulation of the impact of urban non-uniformity on precipitation. Advances in Atmospheric Sciences 33, 783−793.

Steinberg, D.A., Pouyat, R.V., Parmelee, R.W., Groffman, P.M., 1997. Earthworm abundance and nitrogen mineralization rates along an urban-rural land use gradient. Soil Biology and Biochemistry 29, 427−430.

Szlavecz, K., Placella, S.A., Pouyat, R.V., Groffman, P.M., Csuzdi, C., Yesilonis, I.D., 2006. Invasive earthworm species and nitrogen cycling in remnant forest patches. Applied Soil Ecology 32, 54−62.

Szlavecz, K., McCormick, M., Xia, L., Saunders, J., Morcol, T., Whigham, D., Filley, T., Cs, C., 2011. Ecosystem effects of non-native earthworms in Mid-Atlantic deciduous forests. Biological Invasions 15, 1165−1182.

Taylor, G., Tallis, M.M., Giardina, C.P., Percy, K.E., Miglietta, F., Gupta, P.S., Gioli, B., Calfapietra, C., Gielen, B., Kubiske, M.E., Scarascia-Mugnozza, G.E., Kets, K., Long, S.P., Karnosky, D.F., 2008. Future atmospheric CO_2 leads to delayed autumnal senescence. Global Change Biology 14, 264−275.

Trammell, T.L.E., Carreiro, M.M., 2011. Vegetation composition and structure of woody plant communities along urban interstate corridors in Louisville, KY, U.S.A. Urban Ecosystems 14, 501−524.

Trammell, T.L.E., Carreiro, M.M., 2012. Legacy effects of highway construction disturbance and vegetation management on carbon dynamics in forested urban verges. In: Lal, R., Augustin, B. (Eds.), Carbon Sequestration in Urban Ecosystems. Springer: Springer Science + Business Media, pp. 331−352.

Trammell, T.L.E., Carreiro, M.M., August 12, 2015. Nitrogen resorption of urban trees in high N deposition forests and fertilized lawns. In: Presented at the 100th Annual Meeting of the. Ecological Society of America, Baltimore, MD, 2015.

Trammell, T.L.E., D'Amico, V., Avolio, M.L., Mitchell, J.C., Moore, E., 2019. In review. Non-native plant invasion is a stronger driver of vegetation composition in temperate deciduous forests than urbanization. *Journal of Ecology* (in revision).

Trammell, T.L.E., Pataki, D.E., Cavender-Bares, J., Groffman, P.M., Hall, S.J., Heffernan, J.B., Hobbie, S.E., Morse, J.L., Neill, C., Nelson, K.C., 2016. Plant nitrogen concentration and isotopic composition in residential lawns across seven US cities. Oecologia 181, 271−285.

Trammell, T.L.E., Ralston, H.A., Scroggins, S.A., Carreiro, M.M., 2012. Foliar production and decomposition rates in urban forests invaded by the exotic invasive shrub, *Lonicera maackii*. Biological Invasions 14, 529−545.

Trammell, T.L.E., Tripler, C.E., Carper, S.C., Carreiro, M.M., 2017. Potential nitrogen mineralization responses of urban and rural forest soils to elevated temperature in Louisville, KY. Urban Ecosystems 20, 77−86.

Trammell, T.L.E., Day, S., Pouyat, R.V., Rosier, C., Scharenbroch, B., Yesilonis, I., 2018. Drivers of urban soil carbon dynamics. In: Lal, R., Stewart, B.A. (Eds.), Urban Soils: Advances in Soil Science. Taylor and Francis Publishing, Boca Raton, FL.

United Nations, 2015. World Population Prospects: The 2015 Revision, Key Findings and Advance Tables. Working Paper No. ESA/P/WP.241.

United States Census Bureau, 2010. Annual Estimates of the Resident Population: April 1, 2010 to July, 2013. US Census Bureau, Population Division, Source. Release date: March 2014. http://www.census.gov/population/metro/data/index.html.

US Department of Agriculture, Natural Resource Conservation Service, 2001. Summary Report: 1997 National Resources Inventory (Revised December 2001). US Department of Agriculture, Natural Resource Conservation Service, Washington, DC.

Valtera, M., Schaetzl, R.J., 2017. Pit-mound microrelief in forest soils: review of implications for water retention and hydrologic modelling. Forest Ecology and Management 393, 40−51.

Vellend, M., 2003. Habitat loss inhibits recovery of plant diversity as forests regrow. Ecology 84 (5), 1158−1164.

Watmough, S.A., Hutchinson, T.C., Sager, E.P.S., 1998. Changes in tree ring chemistry in sugar maple (Acer saccharum) along an urban-rural gradient in southern Ontario. Environmental Pollution 101, 381−390.

Werkenthin, M., Kluge, B., Wessolek, G., 2014. Metals in European roadside soils and soil solution −A review. Environmental Pollution 189, 98−110.

Wong, F., Harner, T., Liu, Q., Diamond, M.L., 2004. Using experimental and forest soils to investigate the uptake of polycyclic aromatic hydrocarbons (PAHs) along an urban-rural gradient. Environmental Pollution 129, 387−398.

Wong, C.S.C., Li, X.D., Thornton, I., 2006. Urban environmental geochemistry of trace metals. Environmental Pollution 142, 1−16.

Wustman, B.A., Oksanen, E., Karnosky, D.F., Noormets, A., Isebrands, J.G., Pregitzer, K.S., Hendrey, G.R., Sober, J., Podila, G.K., 2001. Effects of elevated CO_2 and O_3 on aspen clones varying in O_3 sensitivity: can CO_2 ameliorate the harmful effects of O_3? Environmental Pollution 115, 473−481.

Xu, S., Xu, W., Chen, W., He, X., Huang, Y., Wen, H., 2014. Leaf phenological characters of main tree species in urban forest in Shenyang. PLoS One 9 (6), e99277.

Yesilonis, I.D., Pouyat, R.V., 2012. Carbon stocks in urban forest remnants: atlanta and Baltimore as case studies. pages 103-120. In: Lal, R., Augustin, B. (Eds.), Carbon Sequestration in Urban Ecosystems. Springer.

Yesilonis, I.D., Pouyat, R.V., Neerchal, N.K., 2008. Spatial distribution of metals in soils in Baltimore, Maryland: role of native parent material, proximity to major roads, housing age and screening guidelines. Environmental Pollution 156, 723−731.

Yu, S., Zhu, Y.G., Li, X.D., 2012. Trace metal contamination in urban soils of China. The Science of the Total Environment 421, 17−30.

Zemlyanitskiy, L.T., 1963. Characteristics of the soils in the cities. Soviet Soil Science 5, 468−475.

Zhang, C.S., 2006. Using multivariate analyses and GIS to identify pollutants and their spatial patterns in urban soils in Galway, Ireland. Environmental Pollution 142, 501−511.

Zhu, W., Careirro, M.M., 2004. Temporal and spatial variations in nitrogen transformations in deciduous forest ecosystems along an urban-rural gradient. Soil Biology and Biochemistry 36, 267−278.

Zipperer, W.C., 2002. Species composition and structure of regenerated and remnant forest patches within an urban landscape. Urban Ecosystems 6 (4), 271−290.

Ziska, L.H., Gebhard, D.E., Frenz, D.A., Faulkner, S., Singer, B.D., Straka, J.G., 2003. Cities as harbingers of climate change: common ragweed, urbanization, and public health. The Journal of Allergy and Clinical Immunology 111, 290−295.

Ziska, L.H., Bunce, J.A., Goins, E.W., 2004. Characterization of an urban-rural CO_2/temperature gradient and associated changes in initial plant productivity during secondary succession. Oecologia 139, 454−458.

Ziter, C., Turner, M.G., 2018. Current and historical land use influence soil-based ecosystem services in an urban landscape. Ecological Applications 28 (3), 643−654.