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Chemical, Physical, and Biological Characteristics of Urban Soils

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Abstract

Urban soils provide an array of ecosystem services to inhabitants of cities and towns. Urbanization affects soils and their capacity to provide ecosystem services directly through disturbance and management (e.g., irrigation) and indirectly through changes in the environment (e.g., heat island effect and pollution). Both direct and indirect effects contribute to form a mosaic of soil conditions. In the Baltimore Ecosystem Study (BES), we utilized the urban mosaic as a series of "natural experiments" to investigate and compare the direct and indirect effects of urbanization on soil chemical, physical, and biological properties at neighborhood, citywide, and metropolitan scales. In addition, we compared these results with those obtained from other metropolitan areas to assess the effects at regional and global scales and to assess the generality of these results. Our overall results suggest that surface soils of urban landscapes have properties that can vary widely, making it difficult to define or describe a typical "urban" soil or soil community. Specifically, we conclude that (i) urban effects on soils occur at multiple scales; (ii) management effects are greater than environment effects, although environmental effects are more widespread reaching beyond the boundary of most urban areas; (iii) urban landscapes are biologically active in pervious areas and have a high potential for carbon storage and nitrogen retention; (iv) the importance of urban and native factors depends on the property being measured; and (v) cross-city comparisons support in part the biotic homogenization and urban ecosystem convergence hypothesis.

Soils in urban landscapes are generally thought of as highly disturbed and heterogeneous, with little systematic pattern in their characteristics. As such, most studies have focused on human-constructed soils along streets and in

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highly impacted areas (e.g., Craul and Klein, 1980; Patterson et al., 1980; Short et al., 1986a; Jim, 1993, 1998). As a result, "urban soils" have been viewed as drastically disturbed and of low fertility (Craul, 1992). However, observations of entire landscapes have shown that the chemical, physical, and biological response of soils to urban land use is complex and variable, such that soils that are largely undisturbed or of high fertility also have been identified in urban areas (e.g., Schleuss et al., 1998; Hope et al., 2005; Pouyat et al., 2007a).

Unfortunately, the response of soil to urban land use is considered by many soil taxonomists to diverge from natural soil formation, and as a consequence, changes in soil characteristics resulting from urban development have received limited attention in the current U.S. soil taxonomy (Fanning and Fanning, 1989; Effland and Pouyat, 1997; Evans et al., 2000). Recent efforts, however, have made progress in developing a taxonomic system for highly disturbed soils (Lehmann and Stahr, 2007; Rossiter, 2007; International Committee on Anthropogenic Soils, 2007), yet the challenge remains to identify a systematic pattern of soil responses to various disturbances, management activities, and environmental changes that typically occur in urban landscapes.

Although the current U.S. soil taxonomy neglects soils altered by urban land use, by definition the taxonomy states that a soil is "... a collection of natural bodies on the earth's surface, *in places modified or even made by man of earthy materials*, containing living matter and supporting or capable of supporting plants out-of-doors" (Soil Survey Staff, 1975), which suggests that soils of urban landscapes should be considered taxonomically with nonurban soils (Effland and Pouyat, 1997; Pouyat and Effland, 1999). Even without a taxonomic designation, definitions of soils associated with urban and urbanizing landscapes have been proposed in the literature. For example, Craul (1992) modified the definition of Bockheim (1974) and defined urban soil 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." As another definition, Evans et al. (2000) suggested the term *anthropogenic soil*, which places urban soils in a broader context of humanly altered soils rather than limiting the definition to urban areas alone. Similarly, Pouyat and Effland (1999) and more recently Lehmann and Stahr (2007) more broadly defined urban soils to include not only those soils that are physically disturbed (e.g., old industrial sites and landfill) but also those that are undisturbed and altered by urban environmental change (e.g., temperature or moisture regimes).

In this chapter, we report on the broad array of effects of urban land use on the physical, chemical, and biological responses of soil, drawing from our research in the Baltimore Ecosystem Study (BES, http://beslter.org, verified 19 Feb. 2010), one of two urban Long Term Ecological Research (LTER) sites funded by the National Science Foundation, as well as research reported in the literature. We begin by introducing soil as the "brown infrastructure" of human settlements and discussing the ecosystem services provided by soils to the inhabitants of urban and exurban areas. We next provide a conceptual framework to incorporate the wide-ranging spatial and temporal effects of urban land uses on soil formation and describe the "urban soil mosaic" as a template to study urban soils. We conclude by presenting case studies of soil responses measured at various scales in landscapes altered by urban and exurban development. We include responses related to the physical, chemical, and biological (i.e., soil fauna and nutrient cycling) characteristics of soils with the ultimate goal of identifying a systematic pattern of these responses with respect to urban land use.

Importance in Urban Ecosystems

Soils form the foundation for many ecological processes and interactions, such as nutrient cycling, distribution of plants and animals, and ultimately location of human habitation (Brady and Weil, 1999). Although soils in urban and urbanizing landscapes are predominately altered by human activity, they provide many of the same ecosystem services as unaltered soils (Effland and Pouyat, 1997). As such, soils can function in urban landscapes by reducing the bioavailability of pollutants, storing carbon and mineral nutrients, serving as habitat for soil and plant biota, and moderating the hydrologic cycle through absorption, storage, and supply of water (Bullock and Gregory, 1991; De Kimpe and Morel, 2000; Lehmann and Stahr, 2007; Pouyat et al., 2007a,b).

In providing these services, soil plays a unique role as the brown infrastructure of urban ecological systems, much in the same way urban vegetation is thought of as green infrastructure (Pouyat et al., 2007a; Heidt and Neef, 2008). Whereas green infrastructure provides services attributed to vegetation, such as the moderation of energy fluxes by tree canopies (Akbari, 2002; Heidt and Neef, 2008), brown infrastructure provides ecosystem services attributed to soil, such as storm water infiltration and purification, and as a support medium for built structures (De Kimpe and Morel, 2000; Lehmann and Stahr, 2007; Pouyat et al., 2007b).

Habitat and Medium for Animals and Plants

On regional and global scales the conversion of native habitats to urban land uses greatly contributes to local extinction rates of plant and animal species (McKinney, 2002; Williams et al., 2009). Exasperating the effect of habitat loss, urban areas are epicenters of many introductions of aboveground and belowground nonnative species, some of which have become invasive or important pathogens or insect pests (Lilleskov et al., 2008; McKinney, 2008; Chapter 12, Reichard, 2010, this volume). The extinctions of native species and the naturalization of urban-adapted species have led to assemblages of species novel to urban areas (McIntyre et al., 2001; Korsós et al., 2002; Hornung and Szlavecz, 2003; Williams et al., 2009). The net result is a general pattern of nonnative species increasing and native species decreasing from outlying rural areas to urban centers.

Plants

Even with the depression of native species richness, the overall species richness of plants may be greater in urban than in rural habitats (McKinney, 2008). The higher species richness of plants in urban landscapes is due to the preferences of people and the naturalization of introduced species (Nowak, 2000; Williams et al., 2009). For instance, Nowak (2010) measured Shannon–Weiner Diversity Index values ranging from 3.0 to 3.8 across several cities in the eastern United States. These values are higher than the range of values found for eastern deciduous forests (1.9–3.1, Barbour et al., 1980). Likewise, Hope et al. (2003) found in the Phoenix metropolitan area that plant species richness was greater in developed areas than in the surrounding desert. Moreover, within the urban areas, species diversity was positively correlated to household income.

Therefore, it appears that soils of urban landscapes can support a greater number of plant species than the native soils they replaced, albeit with some species requiring supplements of water and nutrients. Even without supplements, urban soils appear to have sufficient resources to support plant growth. For example, observations of surface soils in Baltimore City have shown that chemical and physical characteristics fall within the range of requirements for most plants (Pouyat et al., 2007a). Specifically, only 10% of the sampled locations had bulk density measurements greater than 1.4 Mg m^{-3}, the level above which root growth is curtailed in silt loam soils. Moreover, concentrations of potassium, magnesium, and phosphorus were in most cases sufficient for plant growth, while concentrations of calcium exceeded the recommended ranges for horticultural plants in the region. The authors concluded that the apparent accumulation of Ca in these soils occurred due to the widespread use of concrete and gypsum as construction materials, which eventually degrade and get redistributed in the landscape (e.g., Lovett et al., 2000; Juknys et al., 2007).

Soil Fauna

Soil biota is an important component of the soil ecosystem, actively contributing to soil formation by altering its physicochemical properties. All major invertebrate taxa are represented in the soil, and in most terrestrial ecosystems the highest species diversity is found in the soil. Many soil taxa are poorly known, yet new species have actually been discovered and described in urban landscapes (Csuzdi and Szlavecz, 2002; Foddai et al., 2003; Kim and Byrne, 2006). Still less is known about the natural history and ecology of soil fauna than animals in the aboveground community, which is particularly true of urban ecosystems because these systems have been studied less than nonurban systems.

Most soil organisms are part of the decomposer food web, so their major ecosystem function is processing detritus and mobilizing nutrients (Chapter 18, Aitkenhead-Peterson et al., this volume). The soil food web is extremely complex and is currently an important focus of soil ecological research (Bardgett, 2005). Many soil ecologists consider the soil food web highly redundant (e.g., Andrèn et al., 1995; Laakso and Setälä, 1999) meaning that species can be replaced without major functional consequences. Whether functional redundancy exists or not in urban soil communities has important implications because of the presence in urban landscapes of nonnative species, which often occur more abundantly there than native species (Lilleskov et al., 2010).

The composition and abundance of urban soil fauna are determined by many interacting factors, both natural and anthropogenic, and will vary by taxon. Factors contributing to high species richness include the mosaic of land-use and cover types typically existing in urban landscapes and the likelihood of the introduction and establishment of nonnative species (McIntyre et al., 2001; Smith et al., 2006; Byrne et al., 2008; Lilleskov et al., 2008). Moreover, the occurrence of soil organisms in novel habitats, such as built structures, greenhouses, and green roofs, adds to the species richness of urban landscapes (Korsós et al., 2002; Schrader and Böning, 2006; Jordan and Jones, 2007; Csuzdi et al., 2008).

The proportion of nonnative species in an urban landscape is highly taxon dependent and varies with geographical region. Studies conducted in the

Baltimore metropolitan area showed that all carrion beetle species (*Silphidae*) are native (Wolf and Gibbs, 2004), while all terrestrial isopods (*Oniscidea*) were introduced (Hornung and Szlavecz, 2003). In addition, the proportion of nonnative earthworms in the Baltimore metropolitan area is roughly 50% (Szlavecz et al., 2006), and in the New York City metropolitan area no native earthworms occur (Steinberg et al., 1997). As a result, urban soils have fundamentally different soil faunal communities, with a higher proportion of introduced species when compared with their native soil counterparts (e.g., Spence and Spence, 1988; Pouyat et al., 1994; Bolger et al., 2000; McIntyre et al., 2001; Connor et al., 2002).

The successful adaptation of soil fauna to urban environmental conditions involves many physiological and behavioral traits. Individual organisms can cope with environmental stress through behavioral (e.g., migration, shift in food preferences, seeking more favorable microhabitats) or physiological (e.g., regulating absorption and storage of heavy metals) mechanisms (Ireland, 1976; Alikhan, 2003; Lev et al., 2008). Early studies of soil fauna inhabiting urban landscapes focused on pollution tolerance, particularly to contamination by heavy metals. A wealth of information is available on this topic, and results thus far have revealed a complex relationship between levels of contamination and physiology, behavior, and life history of these organisms (Gish and Christensen, 1973; Beeby, 1978; Ash and Lee, 1980; Pizl and Josens, 1995). For instance, responses by soil invertebrates to elevated levels of metals or other pollutants have varied by taxonomic group (Ireland, 1983; Lee, 1985; Beyer and Cromartie, 1987; Morgan and Morgan, 1993). Moreover, responses by individuals within the same population can vary by age, maturity, season, diet, and genetic differences (Ireland, 1983; Spurgeon and Hopkin, 2000).

In addition to the potential for pollution effects on individual soil organisms, the bioaccumulation of metals or other pollutants in urban soil fauna can subject higher trophic organisms, such as predators, to contaminants. Typically the accumulation of metals is magnified the higher the trophic level (Getz et al., 1977; Hopkin and Martin, 1985). In urban landscapes, the potential for the biomagnification of metals is especially true for predators of earthworms, such as birds, lizards, and mammals (Loumbourdis, 1997; Komarnicki, 2000).

Water Infiltration and Storage

Urban landscapes typically exhibit complex and variable soil drainage patterns and moisture regimes (Chapters 14 [Burian and Pomeroy, 2010] and 15 [Steele et al., 2010] this volume). This complexity is a result of a combination of urban factors that either increase or decrease the content of water in soils. For example, highly impacted urban soils often exhibit hydrophobic soil surfaces, surface crust formation, and high bulk densities that restrict infiltration rates (Craul, 1992). Moreover, under urban environment conditions such as heat stress, soil water is more likely to be depleted through higher rates of evapotranspiration. By contrast, soils in urban areas often are irrigated and have abrupt textural and structural interfaces that can restrict drainage resulting in higher soil water contents (Craul, 1992; Pouyat et al., 2007b). Additionally, urban landscapes often have surface drainage features that concentrate water flows (Tenenbaum et al., 2006; Pouyat et al., 2007b; Chapters 14 [Burian and Pomeroy, 2010] and 15 [Steele et al., 2010] this volume). Further complicating these effects are belowground infrastructures that can alter soil water through pressurized potable water distribution systems, which can leak water into adjacent soils, or through storm and sanitary water systems, which can drain soils of water through fractures in pipes (Band et al., 2004; Wolf et al., 2007).

The net result of these differing effects on soil water resources and ultimately the water cycle is unclear and represents an opportunity for future research (Mohrlok and Schiedek, 2007). Preliminary results in the Baltimore Ecosystem Study have revealed differences in soil moisture regimes between types of vegetation cover. In one study, a comparison between nonirrigated residential lawns and an adjacent remnant forest showed that the lawn soils had higher moisture levels at a 10-cm depth than the forest soil during the growing season. However, there were no differences between the two patch types after leaf drop. Moisture differences in the summer were apparently due to higher transpiration rates of the broad-leaved trees (Pouyat et al., 2007b). At a greater soil depth a more complex relationship was revealed in the long-term monitoring of forest and grass plots in the Baltimore metropolitan area. There were no consistent differences in soil moisture between grass and forest plots, although over the entire period from 2001 to 2005, moisture was significantly ($p < 0.05$) higher in grass than forest plots at a 50-cm depth (Groffman et al., 2009). There was an opposite pattern at a 30-cm depth, with significantly higher moisture in forest plots in 2001, 2002, and 2003.

In addition to the importance of cover, studies conducted at the scale of a watershed have shown impervious surfaces and soil disturbances can disrupt the relationship between topography and soil drainage that typically exist in unaltered landscapes. For example, in the Baltimore metropolitan area, Tenenbaum et al. (2006) compared a suburban watershed with a similar sized forested watershed and showed that the developed watershed lacked the typically strong relationship between topographic position and soil moisture. The authors concluded the poor relationship was primarily due to low infiltration and high runoff rates in the suburban watershed. Consequently, it may not be accurate to infer soil moisture or drainage when using topographic maps of urban landscapes.

In addition to a disconnect between soil moisture and topography, urban landscapes may have intact soils that are disturbed only at the surface and thus exhibit a restricted rate of infiltration. As a result, these soils do not hydrologically function as the same soil described in a nonurban context (Pitt and Lantrip, 2000). The potential of urban factors to restrict infiltration rates even in relatively intact soils is of particular importance because of the marked effect soil infiltration can have on stream flows during storm events (e.g., Holman-Dodds et al., 2003).

Sink for Trace Metals

For urban soils, elevated heavy metal concentrations are almost universally reported, although often with high variances (e.g., Wong et al., 2006: Table 7–1). Most of the heavy metal sources in urban landscapes have been associated with roadside environments (Van Bohemen and Janssen van de Laak, 2003; Zhang, 2006; Yesilonis et al., 2008), interior and exterior paint (Mielke, 1999), stack emissions (Govil et al., 2001; Walsh et al., 2001; Kaminiski and Landsberger, 2000), management inputs (Russell-Anelli et al., 1999), and industrial waste (Schuhmacher et al., 1997). Moreover, as heavy metals are emitted into urban environments, they may accumulate onto built surfaces and in the soil, which will vary depending on the source (Mielke, 1999). Cook and Ni (2007) found that relatively heavy

Table 7–1. Surface soil heavy metal elemental means of data reported in the literature.

Author	City	N	Extraction solution	Cd	Co	Cr	Cu	Mn	Ni	Pb	Zn
				mg kg^{-1}							
Aelion et al. (2008)†	South Carolina	60	HNO_3/HCl	–‡	–	7	3	86	2	12	–
Ikem et al. (2008)	New Madrid County, Missouri	62	HF, $HClO_4$, HNO_3	1.6 ± 1.38	10 ± 5.0	25 ± 10.6	18 ± 16.7	298 ± 172	16 ± 4.8	49 ± 39.8	96 ± 117
Kay et al. (2008)	Chicago, Illinois	57	USGS procedures	–	11 ± 3.7	71 ± 49.6	151 ± 373	584 ± 511	36 ± 23.5	395 ± 494	397 ± 411
Lee et al. (2006)	Hong Kong	236	strong acid digestion	0.36 ± 0.16	4 ± 1.6	18 ± 5.92	16 ± 22.6	–	4 ± 2.5	88 ± 62	103 ± 91.3
Odewande and Abimbola (2008)	Ibadan, Nigeria	106	HNO_3/HCl	8.4 ± 19.8	–	–	47 ± 44.1	1098 ± 522	20 ± 14.6	95 ± 127	228 ± 366
Ruiz-Cortes et al. (2005)†	Seville, Spain	52	three-step BCR, HNO_3/HCl	2.89	–	38	55	391	21	156	120
Tume et al. (2008)	Talcahuano, Chile	7	HNO_3/HCl	–	–	38 ± 16.7	–	–	23 ± 6.4	35 ± 43.3	333 ± 364
Yesilonis et al. (2008)1	Baltimore, Maryland	122	HNO_3/HCl	1.06	15	72	45	472	27	231	141
Zhang (2006)	Galway, Ireland	166	HF, $HClO_4$, HCl, HNO_3	–	6 ± 2.4	33 ± 16.3	33 ± 25.5	674 ± 780	21 ± 12.4	78 ± 72	99 ± 68.8
Zhao et al. (2007)	Wuxi, China	102	$HN0_3$/HCl, 3:l	0.14 ± 0.09	–	59 ± 16.2	40 ± 18.2	–	–	47 ± 25.8	113 ± 68.4
Zheng et al. (2008)	Beijing, China	773	HNO_3 and H_2O_2	0.15 ± 0.11	–	36 ± 13.9	24 ± 23.7	–	28 ± 8.7	29 ± 10.3	66 ± 29.8

† No measure of variation given (i.e., standard deviation).

‡ A dash (–) indicates no values were reported for the element.

particles in lead aerosols are deposited on or near roads, while relatively light particles are carried around structures by air currents. Particles of intermediate mass are carried by air currents and have an affinity for structural surfaces.

Soil acts as a sink for heavy metals through sorption, complexation, and precipitation reactions (Yong et al., 1992). These retention mechanisms are regulated by organic matter, pH, cation exchange capacity, and oxides of a soil. Therefore, the heavy metals that reach a soil will vary in their availability to plants, soil fauna, and humans on the basis of how these characteristics spatially vary in the urban landscape. Soils with relatively high amounts of organic matter and oxides and neutral to alkaline pH will generally have both a lowered availability to plants and animals and a lowered leaching rate to groundwater of heavy metals (Pizl and Josens, 1995; Brown et al., 2003). However, when soils are physically disturbed and lose organic matter and base elements, they lose their capacity to bind metals, resulting in an increase in the availability and mobility of metals (Farfel et al., 2005).

Retention and Storage of Carbon and Nitrogen

The most obvious impact on the retention and storage of carbon and nitrogen in urban landscapes is from physical disturbances of soil. Large volumes of soil are typically disturbed during construction, and disturbances continue to occur at finer scales once people inhabit the landscape. Using data collected at a commercial development site in the Baltimore metropolitan area (McGuire, 2004) Pouyat et al. (2007b) estimated that the amount of soil C that was disturbed during a development project of 2600 m^2 in area was roughly 2.7×10^4 kg. What happens to the pool of C disturbed during construction projects has not been reported in the literature to our knowledge. However, it is known that several factors are important to C retention, including the amount of C lost through erosion from the site, the amount of organic rich surface soil that is stockpiled or sold as topsoil, the proportion of the total C pool that is readily oxidized, and the amount of organic C buried during the grading process. These factors should also play a role in the loss of N as either nitrate or nitrous oxide.

The amount of C stored in soil over time, or C sequestered, is a balance between C input through net primary productivity (NPP) and loss through decay (soil heterotrophic respiration), both of which are controlled by environmental factors, including soil temperature and moisture and N availability. Carbon sequestration in urban soils is an important process that helps to mitigate the effects of increased emissions of greenhouse gases into the atmosphere. However, the gain or loss of C from soil can be greatly affected by urban land use and urban environmental change (Pouyat et al., 2002; Lorenz and Lal, 2009). For example, measurements taken in permanent forest and lawn plots of the Baltimore Ecosystem Study have shown that carbon dioxide fluxes from forest soils (a C loss) are increased under urban environmental conditions (Groffman et al., 2006), and the fluxes from managed lawns were as high or higher than the forested sites (Groffman et al., 2009).

Other soil–atmosphere exchanges of greenhouse gases, especially nitrous oxide and methane, are potentially altered by urban land use. Trace gas measurements taken in the Baltimore permanent plots indicate that urban forest and lawn soils have a reduced rate of methane uptake and increased nitrous oxide fluxes in comparison to rural forest soils (Groffman and Pouyat, 2009; Groffman et al., 2009). Likewise, Kaye et al. (2004) found that lawns in Colorado had reduced methane uptake and increased nitrous oxide fluxes relative to native shortgrass steppe. Goldman et al. (1995) found reduced methane uptake in urban versus rural forest patches in the New York City metropolitan area. In all these cases, changes in methane and nitrous oxide fluxes were associated with N inputs such as fertilization and N deposition that typically occur in urban landscapes (Groffman and Pouyat, 2009).

While the potential for losses of C and N in urban landscapes can be high, urban soils have the capacity to accumulate a surprising amount of C and N when compared with agricultural or native soils. Horticultural management efforts (e.g., fertilization and irrigation) tend to maximize plant productivity and soil organic matter accumulation for a given climate or soil type and thus increase the capacity of these soils to store C and N. This is particularly true of lawns, where soils are not regularly cultivated and turfgrass species typically grow through an extended growing season relative to most native grassland, forest, and crop ecosystems (Pouyat et al., 2002; Groffman et al., 2009). Indeed, lawn soils have shown a surprisingly high capacity to sequester C and cycle N (Pouyat et al., 2006; Raciti et al., 2008), although the net effect on C uptake may be somewhat lower if C emissions resulting from management activities are taken into account (Gordon et al., 1996; Pouyat et al., 2009b).

Due to the high amount of N inputs into urban landscapes, there is great interest in reducing exports of nitrate from urban soils to coastal receiving waters that are often N limited. There is particular interest in the ability of urban riparian soils to support denitrification, an anaerobic microbial process that converts nitrate into nitrous oxide and other N gases, thus serving as a sink for N leaching from upland soils. Urban riparian zones tend to be drier and more aerobic than riparian zones in agricultural or forested watersheds and therefore support less denitrification (Groffman et al., 2002, 2003; Chapter 13, Stander and Ehrenfeld, 2010, this volume). However, urban landscapes have areas of saturated soils in novel habitats such as stormwater detention basins and relict wetlands that have been shown to support high denitrification rates (Groffman and Crawford, 2003; Stander and Ehrenfeld, 2009).

Conceptual Framework for Urban Soil Formation

Urban Soil Formation

To increase our understanding of how urban land use alters soil formation and ultimately the characteristics of soils in urban landscapes, Pouyat (1991) suggested the use of the Factor Approach, a conceptual model first proposed by Jenny (1941) to describe the formation of soil at landscape scales. The Factor Approach posits that soil and ecosystem development is determined by a combination of state factors that include climate (*cl*), organisms (*o*), parent material (*pm*), relief (*r*), and time (*t*), where the characteristics of any given soil (or ecosystem), *S*, are the function:

$$S = f(cl, o, pm, r, t) \quad [1]$$

Amundson and Jenny (1991) and Pouyat (1991) proposed that human effects can be incorporated into the factor approach by including a sixth or anthropogenic factor *a*, such that

$$S = f(a, cl, o, pm, r, t) \quad [2]$$

To investigate the relative importance of individual factors, Jenny (1961), Vitousek et al. (1983), and Van Cleve et al. (1991) identified on landscapes "sequences" of soil bodies or ecosystems in which a single factor varies while the other factors are held constant, for example, a chronosequence where age *t* is the varying factor, or a toposequence where *r* is the varying factor. Likewise, Pouyat and Effland (1999) proposed that sequences can be used to investigate the effects of urban land use on soil characteristics and ecosystem processes. For example, when the anthropogenic factor *a* plays a role as in Eq. [2], situations where this factor varies over relatively short distances while the remaining factors are held constant would be considered an "anthroposequence" where:

$$S = f(a)_{cl,o,pm,r,t} \quad [3]$$

Equation [3] represents a sequence, or study design, that can be used to investigate the effects of the *a* factor on a soil at landscape scales (i.e., tens to hundreds of kilometers). An anthroposequence also is suitable for comparing the effects of urban land use on soils at varying temporal scales (Pouyat and Effland, 1999). At one extreme, sequences can be identified where the *a* factor acts interdependently with the other factors for time scales in which pedogenic processes take place, and at the other extreme, sequences can be identified in which the *a* factor operates independently of the other factors (Fig. 7–1). For example, if the soil is disrupted, as in the case of a site that is graded to build a structure, the impact on that soil occurs independently of the other soil forming factors (Fig. 7–1a). In this case the temporal scale of the urban alteration is much shorter than the time frame in which most natural pedogenic processes operate. Here the material constituting the nonurban soil (*S* in Eq. [1]) predated the "new" modified soil (S2) and thus is considered the "new" parent material, so that

$$S2 = f(a, cl, o, S, r, t) \quad [4]$$

Essentially, there is a new time zero from which pedogenesis takes place. Much in the same way that the till remaining from a retreating glacier is considered parent material, in an urban example transported material used to fill in a low-lying area

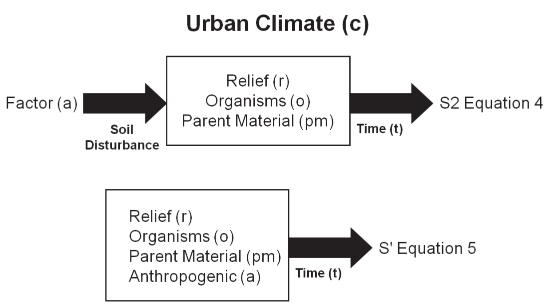

Fig. 7–1. Schematic representation of the temporal effects of the anthropogenic factor (*a*) on soil formation and the relationship of *a* with the natural soil forming factors; climate (*c*), relief (*r*), organisms (*o*), parent material (pm). (A) The anthropogenic factor works at a time frame independently of the other factors. (B) The anthropogenic factor works at a time frame that is interdependent of the other factors (modified from Pouyat and Effland, 1999).

is considered parent material (e.g., Short et al., 1986b). By contrast, factors of soil formation are interdependent, particularly in cases where urban effects occur on time scales similar to soil formation (usually tens to thousands of years, Fig. 7–1b). Under such conditions, interactions with other factors are more likely (Pouyat and Effland 1999; Pickett and Cadenasso, 2009; Pavao-Zuckerman, 2008). Here the profile of the soil remains largely intact, although various chemical, and to a lesser degree physical, properties may be altered through interactions between urban environmental factors and soil formation. Examples of when the *a* factor is interdependent with other soil forming factors include changes in water table depth (Groffman et al., 2003) and changes in soil temperature regimes (Savva et al., 2010) that typically occur in urban landscapes.

Even when a soil in an urban landscape is morphologically similar to a pre-existing condition, the soil may be functionally altered, such as when surface compaction inhibits the ability of an otherwise intact soil to infiltrate water (Pitt and Lantrip, 2000). In these circumstances, the soil parent material (*pm*) remains constant, but for a soil with an altered function (S'), such that

$$S' = f(a, cl, o, pm, r, t) \qquad [5]$$

Therefore, there are two extreme circumstances, differentiated by temporal scales, in which anthroposequences can be defined and used for investigation of urban soils (Pouyat and Effland, 1999):

First, where

$$\mathrm{S2} = f(a_n)_{cl,o,S,r,t} \qquad [6]$$

direct comparisons of different soil disturbances or management regimes associated with urban landscapes (a_1 vs. a_2 and so on) are possible between soils developing under similar environmental situations or parent materials of various origins. Moreover, soil formation can be compared among similar disturbance types along a chronosequence (e.g., Scharenbroch et al., 2005), where

$$\mathrm{S2} = f(t_n)_{a,cl,o,S,r} \qquad [7]$$

In both cases, there is the opportunity to study and control factors (e.g., parent material or management regime) in the early stages of soil development that follows an urban disturbance (Evans et al., 2000).

Second, where

$$S' = f(a_n)_{cl,o,pm,r,t} \qquad [8]$$

effects on soil formation of various urban environmental factors (a_1 vs. a_2 and so on) can be studied over similar soil types, e.g., forested soils along urban–rural environmental gradients (Pouyat et al., 1995; Carreiro et al., 2009). Such comparisons will be useful in delineating threshold responses of soil properties to deposition of atmospheric pollutants (e.g., Pouyat et al., 2008), or gradual changes in soil properties due to changes in temperature regimes (e.g., Savva et al., 2010).

In the first case (Eq. [6–7]), we consider the effects under these circumstances as direct; in the second case (Eq. [8]) we consider them as indirect. The following sections use the above conceptual framework to differentiate direct from indirect effects and the likely expression of these effects spatially and temporally in urban landscapes.

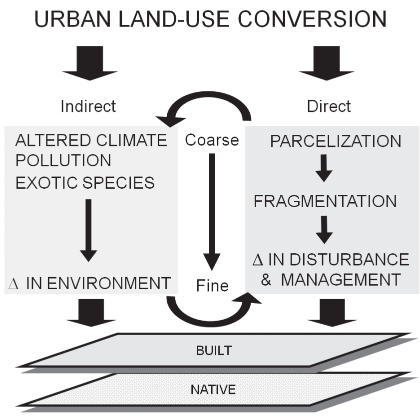

Fig.7–2. Conceptual diagram of the effect of urban land use conversion on native soils. As landscapes are urbanized, natural habitats are increasingly fragmented as parcels of land ownership become smaller. During this process humans introduce novel disturbance and management regimes that impact soil formation; these are known as *direct effects* (arrows on right). Concurrently there is a change in the environmental conditions in which soil formation takes place; these are *indirect effects* (arrows on left). The overlapping of the direct and indirect effects of anthropogenic factors on the native soil results in the "urban soil mosaic." (Modified from Pouyat et al., 2003 and Pouyat et al., 2007b.)

Urban Soil Mosaic

As land is converted to urban uses, direct and indirect factors affect soil chemical, physical, and biological characteristics (Fig. 7–2). Direct effects include those typically associated with urban soils, such as physical disturbances, incorporation of anthropic materials, and burial or coverage of soil by fill material and impervious surfaces (Craul, 1992; Jim, 1998; Schleuss et al., 1998; Galbraith et al., 1999). For most urban landscapes, these types of impacts are more pronounced during rather than after the land development process. Urban development of land typically includes clearing of existing vegetation, massive movements of soil, and building of structures. The extent and magnitude of these initial disturbances is dependent on topography (e.g., McGuire, 2004), infrastructure requirements, and other site limiting factors.

Soil management practices, such as fertilization and irrigation, that are introduced after the initial development disturbance are also considered direct effects. The spatial pattern of disturbance and management practices is largely the result of *parcelization*, or the subdivision of land by ownership, as landscapes are developed for human settlement. The parcelization of the landscape creates distinct parcels with characteristic disturbance and management regimes that will affect soils through time. The net result is a mosaic of soil patches, which will vary in size and configuration dependent on human population density, development patterns, and transportation networks, among other factors (Plate 7–1; see color images section). As mentioned in the previous section, these primarily physical disturbances often lead to "new" soil parent material from which soil develops (Eq. [4], Fig. 7–1a).

Indirect effects related to urban land use change involve changes in the abiotic and biotic environment, which can affect undisturbed soils. In our conceptual framework these effects work at temporal scales in which natural soil formation processes are at work (Eq. [8], Fig. 7–1b). Urban environmental factors include the urban heat island (Mount et al., 1999, Savva et al., 2010), soil hydrophobicity (White and McDonnell, 1988; Craul, 1992), introductions of exotic plant and animal species (Steinberg et al., 1997; Ehrenfeld et al., 2001), and atmospheric deposition of pollutants such as N and S (Lovett et al., 2000; Juknys et

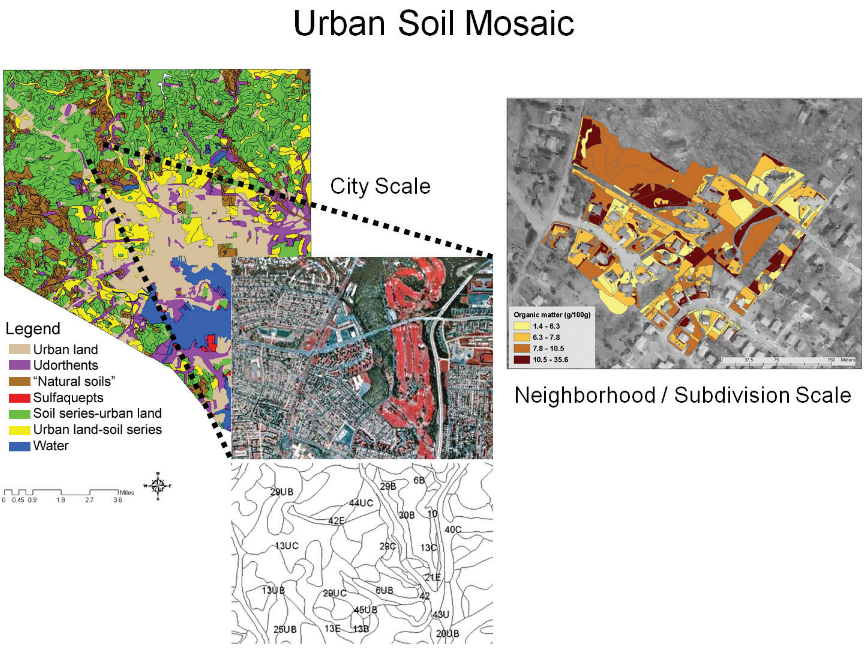

Plate 7–1. The urban soil mosaic shown at multiple scales. Map on the left depicts Baltimore City soil survey units aggregated by native soil series or "natural" soils, urban land, soil series–urban land complex, Udorthents, Sulfaquepts, and urban land–soil series complex (modified from USDA-NRCS, 1998). Center image is an aerial photograph and associated soil map units. Map on the right shows Ecotype Level II polygons for two subdivisions in a suburban neighborhood.

Table 7–2. Fertilizer costs for different land uses in Maryland and turfgrass fertilization rates from the literature.

Land use	Nutrients and other application materials	Managed land	Costs per hectare	Literature values for application rates
		ha		kg N ha^{-1} yr^{-1}
Lawn care firms	$59,124,000	93,688	$631	217–289‡
Golf courses	$10,242,000	13,751	$745	168–239§
Detached single family homes	$248,872,000†	273,173	$911	65¶–120‡

† Costs include mowing equipment, turfgrass nutrients, and other related supplies.

‡ Morton et al. (1988).

§ Klein (1989).

¶ Yesilonis et al., unpublished data, 2004.

al., 2007), heavy metals (Orsini et al., 1986; Boni et al., 1988; Lee and Longhurst, 1992; DeMiguel et al., 1997; Mielke, 1999), and potentially toxic organic chemicals (Wong et al., 2004; Jensen et al., 2007; Zhang et al., 2006).

The net result is a diverse spatial mosaic that represents a variety of soil forming conditions that are useful in comparing the effects of urban land use change on soil characteristics and the distribution of soil fauna (Pouyat et al., 2009a). The spatial heterogeneity of natural soil forming factors may still underlie and constrain the effects of land-use and land-cover change. Additional heterogeneity is introduced from variations in human behavior and social structures that function at multiple scales (Grove et al., 2006; Pickett and Cadenasso, 2009). One example is variation related to fertilization practices among land owners (Table 7–2). The totality of this heterogeneity, both natural and human caused, represents a set of "natural experiments" to investigate the effects of the anthropogenic factor (*a*) on soil formation at different spatial and temporal scales (Eq. [6–8]).

Soil Chemical, Physical, and Biological Responses to Urban Land Use

The urban soil mosaic can be observed at several scales, with each scale of observation revealing a set of soil characteristics that can be related to various anthropogenic and nonanthropogenic soil forming factors (Pouyat et al., 2009a). For example, at the scale of a metropolitan area, the distribution of physically unaltered soil patches tends to increase in density, moving from the highly developed urban core to suburban and rural areas (Effland and Pouyat, 1997). However, these physically unaltered soils and their soil fauna can be affected by changes in environmental factors that are associated with urban land uses at a considerable distance from the urban core (Pouyat et al., 2008; Carreiro et al., 2009). At finer scales, such as the densely populated area of a city or town and the even finer scales of a neighborhood, subdivision, or residence, the characteristics of soils are expected to be more related to anthropogenic factors than to nonanthropogenic ones (Plate 7–1). Moreover, the finer the scale of observation, the more likely a particular human activity or alteration can be related to a specific soil response (e.g., Ellis et al., 2000).

In the next sections, we present several case studies that use the soil mosaic as a suite of "natural experiments" to investigate the response of soil characteristics to urban land use at multiple scales (Pouyat et al., 2009a). We begin with responses of undisturbed soils to environmental factors along urban–rural gradients at the scale of a metropolitan area (Eq. [8]). We then present studies that investigate soil responses at finer scales to different categories of land use and cover, each serving as an "experimental manipulation" of human management efforts, disturbances, and site histories (Eq. [6] and [7]). We finish with a comparison of selected soil characteristics at regional and global scales.

Urban Environmental Effects at the Scale of Metropolitan Areas

Studies using urban–rural gradients suggest 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 atmospheric N inputs over two growing seasons in oak forest stands along an urbanization gradient in the New York City metropolitan area. They found that urban remnant forests received up to a twofold greater N flux than rural forests. More interestingly, the input of N fell off in the stands 45 km from the urban core, which the authors suggest was due to the reaction of relatively small acidic anion aerosols with larger alkaline dust particles high in Ca^{2+} and Mg^{2+} that enabled the precipitation of N closer to the city. Similar results were found for the city of Louisville, KY; the San Bernardino Mountains in the Los Angeles metropolitan area, CA; the city of Oulu, Finland; and the city of Kaunas, Lithuania where N deposition rates, and in some cases S and base cation deposition rates, into urban and suburban forest patches were higher than in rural forest patches (Ohtonen and Markkola, 1991; Fenn and Bytnerowicz, 1993; Bytnerowicz et al., 1999; Juknys et al., 2007; Carreiro et al., 2009).

Evidence of a similar depositional pattern in the form of contents of metals in forest soils was found along urbanization gradients in the New York City, Baltimore, and Budapest, Hungary metropolitan areas. Pouyat et al. (2008) found up to a two- to threefold increase in contents of lead, copper, and nickel in urban forest remnants compared with suburban and rural counterparts. A similar pattern but with greater differences was found by Inman and Parker (1978) in the Chicago, IL metropolitan area, where levels of heavy metals were more than five times higher in urban than in rural forest patches. 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 compact development patterns, exhibited less of a difference between urban and rural remnant forests (Pavao-Zuckerman, 2003; 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 with distance from the urban center to the surrounding rural area. Similarly, Jensen et al. (2007) and Zhang et al. (2006) found significantly higher concentrations of PAHs in surface soils of Oslo, Norway and Hong Kong, China, respectively, than in surrounding rural areas. In all cases the total PAH concentrations were more than twofold higher in urban than in rural areas.

How these pollutants affect soil fauna or soil biological processes is uncertain, but results thus far suggest that the effects are variable and depend on the importance of various urban environmental factors (Pouyat et al., 2007b; Lorenz and Lal, 2009; Carreiro et al., 2009). For example, Inman and Parker (1978) found slower leaf litter decay rates in urban stands that were highly contaminated with Cu (76 mg kg^{-1}) and Pb (400 mg kg^{-1}) compared with unpolluted rural stands, suggesting a negative pollution effect in the Chicago metropolitan area. Similarly, Pouyat et al. (1994) found an inverse relationship between litter fungal biomass and fungivorous invertebrate abundances with heavy metal concentrations along an urbanization gradient in the New York City metropolitan area. By contrast, responses of soil invertebrates along urbanization gradients in Europe were more related to local factors, such as habitat connectivity or patch size, rather than environmental changes occurring along the gradient (Niemelä et al., 2002).

Where heavy metal contamination of soil is moderate to low relative to other atmospherically deposited elements such as N, biological activity may actually be stimulated. For example, decay rates, soil respiration, and soil N transformation increased in forest patches near or within major U.S. metropolitan areas in southern California (Fenn and Dunn, 1989; Fenn, 1991), Ohio (Kuperman, 1999), southeastern New York (McDonnell et al., 1997; Carreiro et al., 2009), and Maryland (Groffman et al., 2006; Szlavecz et al., 2006). However, where excessively high rates of S deposition also occur, several soil biological measurements were negatively affected in forests near an industrial city in northern Finland (Ohtonen, 1994).

In addition to environmental changes related to the inputs of chemicals into soils, urban areas are the foci for many introduced plant and animal species, some of which are invasive and can have large effects on soil processes (Lilleskov et al., 2010). In particular, 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-use change has important implications for soil mediated ecosystem processes (Pouyat et al., 2007b). 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 forested 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). Likewise, plant species invasions can impact C and N dynamics, 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 the shrub *Berberis thunbergii* DC., the tree *Rhamnus cathartica* L., and the grass *Microstegium vimineum* (Trin.) A. Camus (Ehrenfeld et al., 2001; Heneghan et al., 2002; Kourtev et al., 2002).

Land Use and Cover at the City and Neighborhood Scale

Land use and cover can serve as an indicator of disturbance, site history, and management—factors that, as previously mentioned, have the potential to affect soil characteristics. As such, drawing a relationship between soil characteristics and land use and cover in an urban context should advance our understanding of anthropogenic effects of soils. Moreover, if these relationships are systematic, they will be useful in the development of mapping concepts for soils in urban landscapes. Therefore, an important task in urban soil research is to establish at multiple scales coherent relationships between soil characteristics (surface and subsurface) and land use and cover (Pouyat et al., 2007a, 2009a).

As population density increases, the parcels of landownership become smaller in area; thus, management and disturbance occur at finer scales. For this reason, to capture the heterogeneity inherent in the urban soil mosaic, the patches that make up the mosaic require delineation at relatively fine scales (Ellis et al., 2006; Pickett and Cadenasso, 2009). Management can vary significantly in urban mosaics (Table 7–2), which should result in a corresponding response by the affected soils, although the response will be tempered by other anthropogenic factors and the characteristics of the native soil that remain. Therefore, an important question related to the formation of urban soils is the relative importance of human alterations such as landscape management versus the importance of individual natural soil forming factors (i.e., factor *a* versus *pm*, *r*, *o*, *cl*, and *t* in Eq. [2]). We address this question and look for a systematic pattern in characteristics of surface soils in the following sections by comparing the spatial relationships of various soil variables with land use, cover, and the natural soil forming factor (*pm*) at the scale of a city, neighborhood, or subdivision (Plate 7–1).

City Scale

To investigate the relationship between urban land use and soil characteristics at a 0- to 10-cm depth, 130 plots of 0.04 ha were stratified by Anderson Level II land-use and cover classes as part of the Baltimore Ecosystem Study (Nowak et al., 2004; Pouyat et al., 2007a). Results showed a wide range of characteristics among all land-use and cover classes, although a subset of the soil variables measured (P, K, bulk density, and pH) were differentiated by the Anderson classes (Pouyat et al., 2007a). Differences were greatest between classes of land use and cover characterized by intensive land management (lawns) and the absence of management (forests). In particular, concentrations of P and K, both of which are components of most lawn fertilizers, and bulk density, an indirect measure of soil compaction, differentiated the forest from the grass cover plots.

Taking a subset of the forest and lawn cover-type plots and adding a set of agricultural plots, Groffman et al. (2009) measured potential N mineralization and nitrification rates among the different cover types. Relatively large differences were found among the forested, lawn, and agricultural land-use types. Moreover, when these data were compared with data collected in forest soils along an urban–rural gradient in the Baltimore metropolitan area (Szlavecz et al., 2006), differences were much higher between land-use and cover (50-fold difference) than between urban and rural forest remnants (10-fold difference) (Pouyat et al., 2009a). These results suggest that soil management associated with different land uses has a much greater effect on N cycling than environmental factors, such as N deposition and temperature changes, that occur along urban–rural gradients. In addition, Scharenbroch et al. (2005) showed that the differences in N cycling that may occur with respect to cover and management will increase as the soil ages after a site disturbance (i.e., a chronosequence). Research in the Phoenix metropolitan area also showed the importance of urban land use and management on soil N processes. Soils associated with urban land uses had significantly higher inorganic N pools than desert soils (Hope et al., 2005; Zhu et al., 2006), and soils associated with irrigated residential landscapes, or "mesic yards," had higher total N and available P than xeric landscaped yards and nonresidential spaces (Kaye et al., 2008).

Unlike N, P, or K, heavy metals such as Pb, Cu, and Zn did not differentiate the land-use and cover classes used in the Baltimore study, but rather were found to be related to major transportation corridors and the age of housing stock (Yesilonis et al., 2008). A similar relationship was found in Baltimore with an earlier investigation of garden soils (Mielke et al., 1983). Several investigations of other cities have found similar associations of these metals with transportation networks, suggesting the importance of vehicular traffic as a source of Pb, Cu, and Zn in urban environments (Bityukova et al., 2000; Facchinelli et al., 2001; Manta et al., 2002; Madrid et al., 2002; Li et al., 2004; Zhang, 2006). Indeed, as an additive to gasoline, Pb was an important component of automobile exhaust until 1986 in the United States (Mielke, 1999), while Cu and Zn contamination continues to occur from brake emissions and tire abrasion, respectively (Councell et al., 2004; Hjortenkrans et al., 2006).

An interesting and important result of the citywide analysis in Baltimore was the continued importance of parent material that differentiated plots by physiographic province (Atlantic Coastal Plain and Piedmont) for a subset of variables measured, primarily soil texture and the trace elements Al, Mg, V, Mn, Fe, and Ni, which are important constituents of the surface rock types found in the Piedmont province. These results show the continued importance of the effect of parent material on surface soils in urban landscapes (Pouyat et al., 2007a).

Neighborhood Scale

To relate soil characteristics at finer scales in the urbanized areas of the Baltimore metropolitan area, we delineated patches with higher categorical resolution at the scale of a neighborhood or subdivision and individual parcel. At this scale, soil responses can be related to individual patches with specific site histories and activities of individual land managers (Pouyat et al., 2009a). Two neighborhoods in the Baltimore metropolitan area, one medium-density (suburban) and the other high-density (urban) residential, were delineated using high resolution ecotope mapping (Ellis et al., 2006). The suburban neighborhood was situated just outside the city boundary and had households with a significantly higher mean income than households in the urban neighborhood. Similar to the citywide analysis described earlier, 80 plots of 0.04 ha were randomly stratified, but this time by ecotope classes of cover and use delineated in each neighborhood. Results showed that lower concentrations of Pb and Cu were found in the suburban neighborhood than in the urban one, and by contrast to the citywide analysis, these elements varied by use and cover classes delineated at a finer resolution, with the highest concentrations occurring in urban vacant and disturbed lots. Moreover, the proportion of plots with concentrations exceeding the USEPA's soil Pb screening level of 400 mg kg^{-1} was more than 16% in the urban neighborhood, while none of the plots in the suburban neighborhood exceeded the USEPA standard for Pb (I.D. Yesilonis and R.V. Pouyat, unpublished data, 2007).

Unlike the neighborhood-scale analysis, Pb concentrations were examined at the parcel level and were found to be consistent with the citywide results. A total of 60 residential properties in Baltimore City were intensively sampled for total Pb using field portable X-ray fluorescence. Thirty percent of sites had average Pb values that exceeded the USEPA screening level while 53% of sites exhibited lead

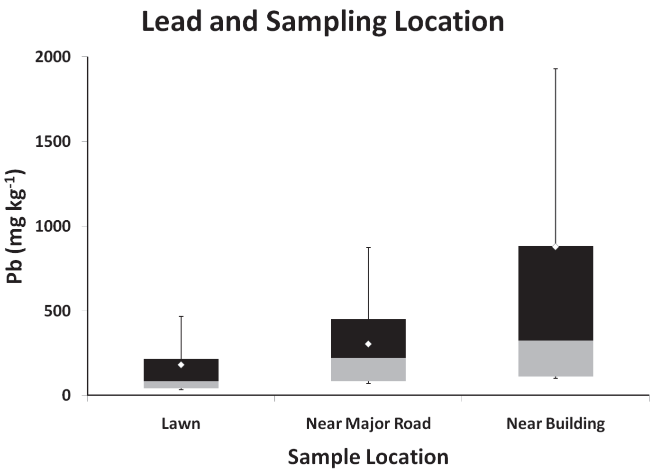

Fig. 7–3. Mean values (white diamonds) and error bars representing the lowest and highest values falling within 1.5 times the interquartile range. Observations outside the error bars are not shown; however, they were not removed from the dataset. Samples were collected from residential lots in Baltimore City. The "lawn" classification represents areas of the lawn not adjacent to a major roadway or building. The "near major road" classification represents samples closest to a major roadway (defined as the primary and secondary roads in the TIGER classification). The "near building" classification represents samples collected directly adjacent to a built structure (Schwarz, 2010).

levels >400 mg kg^{-1} in at least one area of the yard. The highest lead levels were found near buildings and major road networks (Fig. 7–3). Elevated lead levels were observed next to buildings regardless of building type, including brick and wood frame structures. Housing age was an important predictor of soil lead levels with none of the structures built after the ban on lead-based paint and leaded gasoline exhibiting soil lead levels above the EPA limit (Schwarz, 2010).

Using a subset of sample locations in the suburban neighborhood, soil concentrations of Ca, K, Mg, P, and organic matter were compared with N fertilizer rates applied by homeowners (Table 7–2). Both Ca and Mg concentrations showed a significant ($p < 0.01$) indirect response to fertilizer rates ($r^2 = 0.96$ and 0.65, respectively), while organic matter, K, and P did not show a trend. Similarly, springtail (Collembola) and mite (Acari) densities were indirectly related to fertilizer applications in the same neighborhood (Fig. 7–4). However, when we compared two subdivisions within this neighborhood that were built 10 yr apart, P and organic matter concentrations were significantly higher in the older subdivision (Fig. 7–5a). In addition, a comparison with an adjacent forest patch showed that both subdivisions had higher concentrations of P and K, which was consistent with the citywide results (Fig. 7–5b). Again, comparisons at a finer scale (neighborhood vs. citywide) enabled the association between an anthropogenic factor, in this case a management effect (N fertilization), on a soil response (Ca and Mg concentrations and mesofauna abundances), but also verified a relationship at a citywide scale between an intensively managed soil (lawn) versus an unmanaged forest soil.

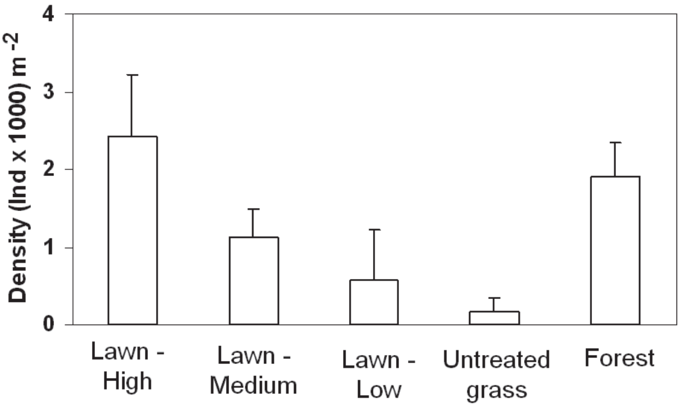

Fig. 7–4. Mean (SE) springtail (Collembola) densities by lawn maintenance level and for a nearby forest patch of a suburban neighborhood in Baltimore County, Maryland. Lawn maintenance level is based on number of times the lawn was fertilized during the growing season (K. Szlavecz, unpublished data, 2006).

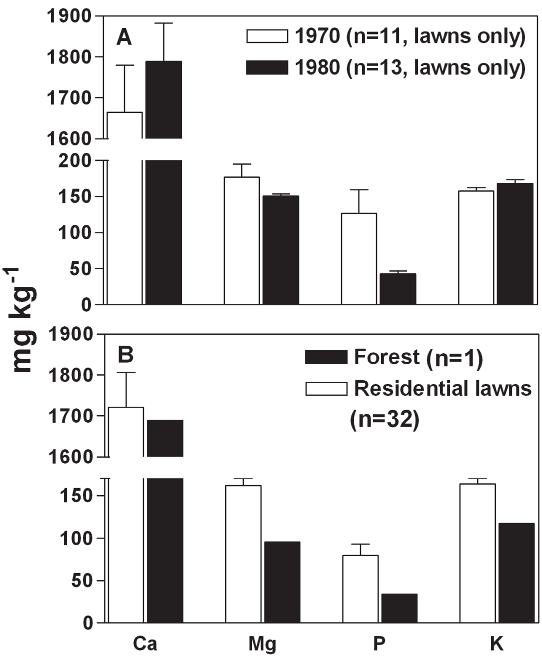

Fig. 7–5. Mean (SE) concentrations of Ca, Mg, P, and K for soils sampled to the 0- to 5-cm depth in a suburban neighborhood in Baltimore County, Maryland: (A) by year of development (1970 and 1980); (B) by land use and cover (Forest and Lawn) (I.D. Yesilonis, unpublished data, 2005).

Comparisons of Urban Soil Characteristics at Regional and Global Scales

A central principle in urban ecological theory presupposes that anthropogenic drivers will dominate natural drivers in the control of ecosystem response variables (Alberti, 1999; Kaye et al., 2006). If this assumption were true, it follows that ecosystem responses to urban land-use change should converge, relative to the native systems being replaced, at regional and global scales—this is termed the *urban ecosystem convergence hypothesis* (Pouyat et al., 2003), and with respect to plant and animal species assemblages, *biotic homogenization* (McKinney and Lockwood, 2001; McKinney, 2006).

The convergence hypothesis suggests that ecosystem responses, like soil response variables, will converge across regional and global scales as long as the anthropogenic drivers (i.e., management, disturbance, and environmental change) dominate over natural soil forming factors like topography and parent material (i.e., *r* and *pm* in Eq. [2]). The result is that soil characteristics will be more similar across urban landscapes at regional and global scales than the native soils they replaced (Pouyat et al., 2008). The biotic homogenization hypothesis suggests that urban land-use change results in local extinctions of regional soil fauna and that the movements of humans across geographical boundaries help facilitate the establishment and spread of urban adapted, or synanthropic, soil fauna. These synanthropic species thrive in urbanized landscapes, resulting in a high degree of similarity in species composition across regional and global scales. While biogeography theory suggests that community similarity should decrease with distance in "natural habitats" (Nekola and White, 1999), in human-dominated systems the decrease in similarity should be less or absent (McKinney, 2006). To address these hypotheses, in the following sections we compare soil responses of heavy metal concentrations, soil C densities, and soil invertebrate abundances to urban factors among several metropolitan areas.

Heavy Metals

To investigate the importance of urban environmental factors (*a*) versus parent material (*pm*) in Eq. [2] on soil chemistry, Pouyat et al. (2008) compared 15 soil response variables measured in remnant forests (deciduous hardwood) among urbanization gradients in the Baltimore, New York, and Budapest metropolitan areas. These metropolitan areas differed in population densities, size of area, and transportation systems. In the case of New York, the forest patches were situated on surface geology of the same type (Pouyat et al., 1995) and thus approximated an anthroposequence (Eq. [3]) along an urbanization gradient of 0 to125 km, whereas in Baltimore and Budapest, the surface geology differed along a 0- to 30-km gradient and a 0- to 20-km gradient, respectively (Szlavecz et al., 2006; Pouyat et al., 2008).

The authors found that forest soils are responding to urbanization gradients in all three metropolitan areas, although features of each city (spatial pattern of development, forms of transportation, parent material, and site history) influenced the soil chemical response. The authors suggested that the changes measured resulted from locally derived atmospheric pollution of Pb, Cu, and to a lesser extent Ca, which was more extreme at the urban end of the gradient, but extended beyond the political boundary of each city. In the case of Pb and

Cu, the soil concentrations were highly correlated to traffic volume and density of roads, confirming the importance of vehicle emissions with respect to heavy metal pollution. Moreover, in Baltimore and Budapest the soil chemical response was confounded by differences in parent material found along those urbanization gradients. A similar result showing the importance of parent material was found in regional analyses of pollution effects on soils in Tallinn, Estonia, and the Piedmonte Region of northwestern Italy (Bityukova et al., 2000; Facchinelli et al., 2001). Thus, the comparison of urban–rural gradients across three distinct metropolitan areas suggests that characteristics of the native parent material persist in the surface of urban soils. However, at the same time, the influence of urban environmental factors (deposition of Pb, Cu, and Ca) resulted in similar soil responses across the three metropolitan areas, which also occurred at distances beyond the boundaries of each city.

Soil Carbon

An important characteristic of urban land-use change with respect to the C cycle is the replacement of native cover types with lawn cover (Kaye et al., 2005; Milesi et al., 2005; Golubiewski 2006; Pouyat et al., 2009b). The estimated amount of lawn cover for the conterminous United States is 163,800 $km^2 \pm 35{,}850\ km^2$, which is approximately three times the area of other irrigated crops (Milesi et al., 2005). To manage this lawn cover, roughly half of residential and institutional managers appear to apply fertilizers (Law et al., 2004; Osmond and Hardy, 2004), with some applying at rates similar to or exceeding those of cropland systems, e.g., >200 kg $ha^{-1}\ yr^{-1}$ (Morton et al., 1988; Table 7–2).

Due to the management efforts of landowners and the expansive use of lawn cover, there should be a significant accumulation of soil C in most urban landscapes (Pouyat et al., 2006; Huh et al., 2008). Qian et al. (2003) showed with simulations of the CENTURY model that N fertilization coupled with grass clipping replacement increased soil C accumulations by up to 59% in comparison with sites that were not fertilized and clippings were removed. Likewise, Golubiewski (2006) measured soil C stocks in 13 residential yards of different ages in the semiarid shortgrass prairie of Colorado and found that soil C recovered from the initial development disturbance after 20 yr and exceeded the semiarid prairie soil in 40 yr. Moreover, the effects of lawn care management on soil C accumulation exceeded the effects of other soil forming factors, such as elevation and soil texture.

The amount of effort to manage lawn or turfgrass systems, including the intensity of irrigation or nutrient applications, also should have an effect on soil C accumulation (Huh et al., 2008). Moreover, the effort should reflect the natural constraints on the turfgrass system (Pouyat et al., 2009b). For example, lawns growing in arid areas require more irrigation than lawns growing in temperate climates. Milesi et al. (2005) found that in the absence of irrigation and fertilization most species of turfgrass would not be able to grow and compete with native vegetation in most of the conterminous United States. Therefore, compensatory inputs of water and nutrients required to maintain turfgrass cover should result in a higher accumulation of soil C than in the previous dryland system (e.g., Golubiewski, 2006; Jenerette et al., 2006; Kaye et al., 2008). However, in more temperate regions of the United States, where turfgrasses may grow with minimal supplements, soil C in residential areas should be equivalent to or lower than the native soil, which will most likely be a forest soil (Pouyat et al., 2009b).

To address the potential for lawns and other urban land use and cover types to accumulate soil C, Pouyat et al. (2006) reviewed the literature and found that across different climate and soil types, older residential lawns had surprisingly similar C densities of 14.4 ± 1.2 kg m^{-2} m^{-1} depth. In addition, the authors conducted an analysis of available soil C data for urban and native soils, which showed remnant patches of native vegetation accounting for up to 34% of the stock of soil C of a city. However, when soils beneath impervious surfaces were excluded from the analysis, the estimated soil C densities rose substantially for the urban land-use and cover types, indicating the potential for soils in pervious areas of urban landscapes to sequester large amounts of C. The authors also compared the pre- and post-urban estimates of soil C stocks in six cities and found the potential for large decreases in soil C post-urban development for cities located in the northeastern United States, where native soils have inherently high soil C densities, but in drier climates with inherently low soil C densities, cities tended to have slightly higher soil C densities than the native soil types.

Soil Invertebrates

Biotic homogenization has been tested primarily for large organisms with relatively straightforward taxonomy, such as plants and birds (McKinney, 2006). However, large-scale zoogeographical analysis of invertebrate communities is challenging because of differences in collection techniques, difficulties identifying species, and confusion about nomenclature (Byrne et al., 2008). Investigating the spatial distribution of soil faunal communities along urban–rural gradients has been a focus of urban ecological research (e.g., Pouyat et al., 1994; Pizl and Josens, 1995; Steinberg et al., 1997; Szlavecz et al., 2006 Hornung et al., 2007). To make results from this approach comparable among many locations, a global network (GLOBENET) using standardized sampling methodology was initiated (Niemelä et al., 2000). Although GLOBENET initially focused on ground beetles (Carabidae) (Niemelä et al., 2002; Ishitani et al., 2003), other epigeic (surface-dwelling) arthropod taxa have also been analyzed (Hornung et al., 2007; Vilisics et al., 2007).

Preliminary results suggest that the degree of biotic homogeneity among soil fauna is not uniform and varies by taxa. For instance, international comparisons of urban carabid beetle assemblages at GLOBENET sites have shown a large degree of local differences (Niemelä et al., 2002). Other taxa, such as carrion beetles (Silphidae) have more specialized natural histories, and their species composition is largely determined by the regional species pool and the size and quality of urban forest habitats (Wolf and Gibbs, 2004). By contrast, earthworm species show a high degree of similarity among cities in the United States and Europe due to a high proportion of synanthropic, peregrine species making up earthworm assemblages (Table 7–3). These species have been carried accidentally or deliberately across continents and are now cosmopolitan in their distribution.

Acting against biotic homogenization is the survival of native species in remnant habitat patches or their adaptation to novel environments, as "urban adapters" (Johnston, 2001). The regional species pool varies geographically, and thus the subset adapting to urban environments will vary as well. Often these species are taxonomically similar and ecologically equivalent, resulting in urban vicariance. Two ecologically similar millipede species that have adapted to urban environments illustrate this phenomenon: *Brachyiulus bagnali* Roleman, a species

Table 7–3. The most commonly occurring earthworm species reported from cities surveyed in Europe and Baltimore, MD.†

Species	Baltimore	Budapest	Brussels (1)	Warsaw (2)	Greifswald (3)	Basel (4)	Bonn (5)	Dorsten (6)
Allolobophora chlorotica Savigny		x	x	x	x	x	x	x
Aporrectodea caliginosa Savigny	x	x	x	x	x	x	x	x
Aporrectodea longa Ude		x	x	x	x	x	x	x
Aporrectodea rosea Savigny	x	x	x	x	x	x	x	
Dendrobaena octaedra Savigny	x	x	x	x	x		x	x
Eiseniella tetraedra Savigny	x	x	x	x	x	x		
Lumbricus castaneus Savigny	x		x	x	x	x	x	x
Lumbricus rubellus Hoffmeister	x	x	x	x	x	x	x	x
Lumbricus terrestris L.	x	x	x	x	x	x	x	x
Octolasion cyaneum Savigny		x	x	x	x	x		x
Octolasion lacteum Örley	x	x	x	x	x	x		x
Total number of species	18	19	14	15	15	13	7	15

† References: (1) Pizl and Josens, 1995; (2) Pilipiuk, 1981; (3) Unger, 1999; (4) Glasstetter and Nagel, 2001; (5) Keplin and Broll, 1999; (6) Fründt and Ruszkowski, 1989.

of Continental Europe has a distribution limited to the eastern European city of Budapest, and *B. pusillus* Leach, an Atlantic European species, has been recorded more widely in cities located throughout Europe. *Brachyiulus bagnali*, while limited to Continental Europe is urban adapted and functionally similar to the *B. pusillus*.

Conclusions

Research on soils of urban landscapes has shown that properties of surface soils and their soil fauna can vary widely, making it difficult to define or describe a typical "urban" soil or soil community. Urban soils vary in characteristics depending on the nature and history of disturbance, management regime, and the effect of urban environmental changes. Moreover, the importance of natural soil forming factors (e.g., *pm* in Eq. [2]) continues to influence the spatial distribution of chemical and physical characteristics even when a significant proportion of the land area has been impacted by urban development.

Although urban landscapes are highly altered by development and human activities, urban soils have exhibited a surprising capacity to support plant growth and soil fauna; as a result they may have relatively high rates of biological activity and richness of species relative to the native systems they have replaced. With this capacity, urban soils have the potential to provide various ecosystem services to inhabitants of urban areas and settlements. However, frequent disturbance and many of the environmental changes occurring in urban landscapes can reduce the capacity of these soils to provide ecosystem services or support a diverse soil community, suggesting the importance of developing sustainable management practices that enhance ecosystem services of urban soils.

Some systematic patterns are emerging from our use of "natural experiments" (e.g., Eq. [5–8]) in the urban soil mosaic. Management, disturbance, site history, and environmental patterns all have been shown to have an impact on the spatial response of surface soils and soil fauna at multiple scales in the urban landscape. While physical and management impacts appear to be more pronounced, environmental factors have the potential to be more widespread, often having influence beyond the political boundary of most urban settlements.

The patterns discovered through our conceptual framework of urban soil formation should be useful in developing mapping concepts of soils in urban landscapes. For example, differences in surface soil properties among land-use and cover types could help determine mapping concepts in soil surveys of urban landscapes. Moreover, those soil properties associated with management, such as fertilizer applications and intensity of use, may be useful as surface diagnostic properties to differentiate human-altered soils associated with urban landscapes. Research on the patterning of the characteristics of soil at and substantially below the surface is needed in metropolitan areas that represent various-sized human settlements, cultural and economic factors, native soil types, and climates to generalize these patterns.

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