CHAPTER 21

Soil Carbon in Urban Forest Ecosystems

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CONTENTS

INTRODUCTION

In the contiguous 48 states of the United States, urban areas increased twofold between 1969 and 1994 and currently occupy 3.5% of the land, or 2.81×10^7 ha (Dwyer et al., 1998). On a global scale, more than 476,000 ha of arable land are converted annually to urban areas (World Resources Institute, 1996). This conversion has the potential to greatly modify soil organic carbon (SOC) pools on regional scales and to a lesser extent on a global scale (Pouyat et al., 2002).

While conversion of native ecosystems to agricultural use and recovery from agricultural use have been well studied, conversions to urban land uses have received little attention. Agricultural uses have generally led to losses of SOC (Mann, 1986; Davidson and Ackerman, 1993). Also, we know that as agricultural practices have been abandoned in previously forested areas, ecosystem

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development leads to a recovery of aboveground C pools over decades (Houghton, et al., 1999; Caspersen et al., 2000). By contrast, land recovery from urban land-use conversions is unlikely, and in cases where urban land is abandoned, recovery should be slow due to poor growing conditions (e.g., Clemens et al., 1984).

Unfortunately, very little data are available to assess whether urbanization leads to an increase or decrease in SOC pools. This paucity of data makes it difficult to predict or assess the regional effects of land-use change on SOC pools in various regions of the world (e.g., Howard et al., 1995; Tian et al., 1999; Ames and Lavkulich, 1999). In this chapter, we discuss the potential for soil disturbances and various urban environmental changes to affect SOC pools and fluxes in urban ecosystems. In addition, we estimate existing SOC pools in urban ecosystems on regional and global scales and compare these pools to various native ecosystems using the limited data available. Finally, we present data to show how SOC pools vary across different land-use types found in urban landscapes.

EFFECTS OF URBANIZATION ON SOIL

As land is converted to urban uses, direct and indirect factors affect SOC pools. Direct effects include physical disturbances, burial or coverage of soil by fill material and impervious surfaces, and soil management inputs (e.g., fertilization and irrigation). The spatial pattern of these disturbances and management practices are largely the result of "parcelization," or the subdivision of land by property boundaries, as landscapes are developed for human settlement. The parcelization of the landscape creates distinct parcels of disturbance and management regimes that will affect the characteristics of soil over time, resulting in a mosaic of soil patches (Figure 21.1). Pouyat and Effland (1999) suggest that physical disturbances often lead to "new" soil parent material from which soil development proceeds. Indirect effects of urbanization involve changes in the abiotic and biotic environment, which can influence the development of intact soils. Unlike direct physical disturbance, changes in environmental factors resulting from urbanization affect soil development at temporal scales (>100 yr) in which natural soil formation processes are at work (Pouyat and Effland, 1999).

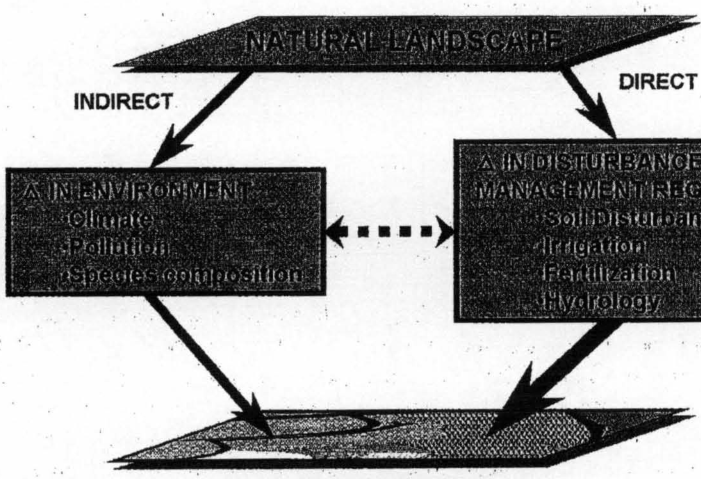

Figure 21.1 Conceptual diagram of the effect of urban land-use conversions on soil. As landscapes are urbanized, there is a change in the environmental conditions in which soil formation takes place. Moreover, humans introduce novel disturbance and management regimes to the landscape. Arrow size indicates the importance of each effect on the resultant mosaic of soil conditions.

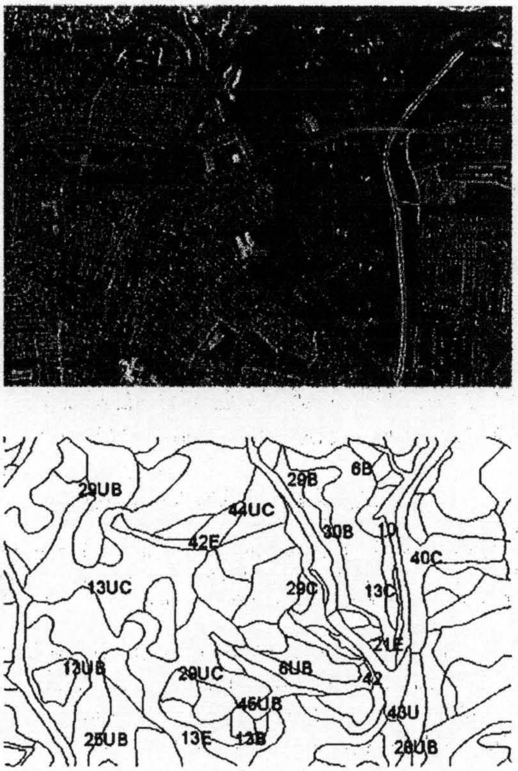

Figure 21.2 An aerial photograph (top) and overlapping soil-map units depicting the land use and spatial complexity of soil in urban landscapes. (Modified from NRCS, Soil Survey of City of Baltimore, Maryland (Soil Survey Report), USDA Natural Resource Conservation Service, Washington, D.C., 1998.)

THE URBAN SOIL MOSAIC

The conversion of agriculture, forest, and grass lands to urban and suburban land uses results in an array of soil patches that range in condition from natural soil profiles to partially disturbed profiles to "made" and covered soils. Overlaid on this mosaic of soil patches are various human activities, such as recreational uses and turf management practices, that can further modify soil profiles and soil characteristics. As the soil mosaic develops, highly variable soil conditions result, the extent of which is dependent on the pattern of development, the range of management regimes in use, and the magnitude and pattern of environmental change (Figure 21.2).

Craul (1992) discussed urban soil variability in the context of vertical and spatial (horizontal) variability on scales of meters to hundreds of meters. Vertical soil variability is observed as soil horizon differentiation or lithologic discontinuities in both undisturbed and disturbed soils. In urban landscapes, vertical and lateral changes in soil horizonation result from human activities such as excavation, backfilling, and the intensity and type of management practices in use (Effland and Pouyat, 1997).

Good illustrations of the spatial variability of soil in urban land can be found in large urban parks. For example, Central Park in New York City is an entirely "made" landscape that has a long history of horticultural, turf, and recreational uses (Cramer, 1993). A soil-testing program has been under way in the park since the early 1980s, and results indicate a wide range in soil properties such as pH and organic-matter concentration (Figure 21.3). Differences in pH and

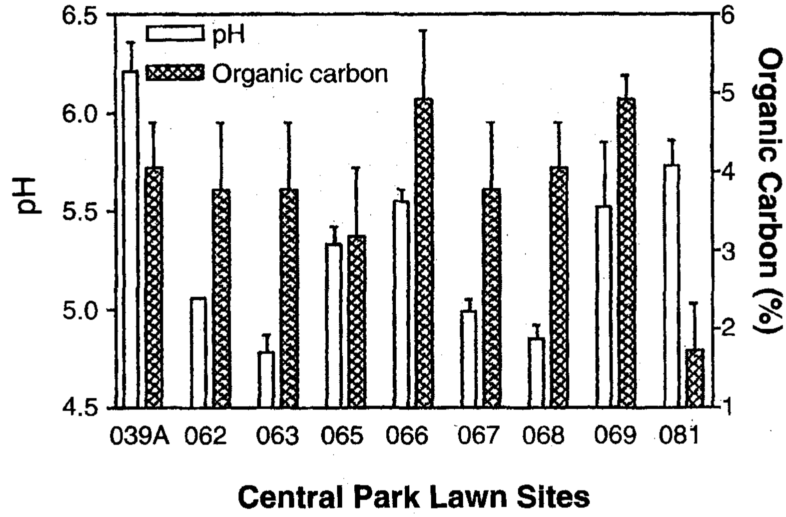

Figure 21.3 Mean (+ SE) pH and percentage of organic matter for lawn areas in Central Park, New York City. Each bar is the mean of two composite samples taken at different times in October 1987. Results for site 062 are from a single sample. (From Kruzansky, R., unpublished data. With permission.)

organic-matter concentration in the lawn areas of the park can be attributed to spatial differences in soil imported for landscape restoration projects and to differences in and intensity of park use. Likewise for an urban park in Hong Kong, differences in SOC ranged more than fourfold between minimally and highly maintained grass areas (Jim, 1998). Similar to the variability found in large urban parks, soils situated in urban, suburban, and rural landscapes in Nigeria showed greater and less-predictable spatial variation of soil properties in the urban than in the rural land uses (Gbadegesin and Olabode, 2000).

Based on studies in metropolitan areas of North America, Pouyat and Effland (1999) suggest that the distribution of unmodified soil patches typically increases in density as one moves from the highly developed core to suburban and rural areas. In the New York City metropolitan area, several researchers have used this urban development pattern by establishing a transect from the Bronx, New York, to rural western Connecticut to study the influence of an urban-rural environmental gradient on undisturbed forest soils (McDonnell et al., 1997). Along this transect, population density, percentage impervious surface, and automobile traffic volume were significantly higher at the urban compared with the rural end of the gradient (Pouyat et al., 1995; Medley et al., 1995). An analysis of these factors indicated that urban and suburban environments have the potential to significantly affect soil characteristics. Higher heavy metals (Pb, Cu, Ni), organic matter, and total salt concentrations, and slightly more-acidic soil solutions were found in the surface 10 cm of forest stands toward the urban end of the gradient. Moreover, these characteristics correlated significantly with measures of the intensity of urban land use surrounding individual forest patches (Pouyat et al., 1995).

CHANGES IN ENVIRONMENT

As we have described, studies of undisturbed forest soils along urban-rural gradients suggest that urban and suburban environments can have significant effects on soil characteristics. Below we discuss various environmental factors associated with urban and suburban landscapes that have the potential to affect SOC fluxes and pools.

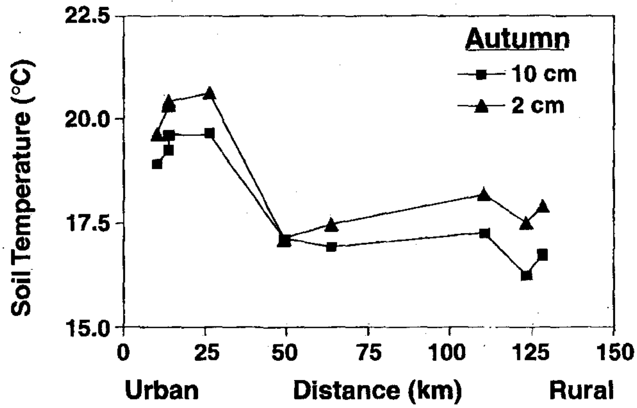

Figure 21.4 Mean soil temperatures (°C) of forest soils at 10 and 2 cm in depth along an urban-rural gradient in the New York City metropolitan area in autumn 1990. Each point represents four temperature measurements on the same day. (Modified from Pouyat, R.V., Soil Characteristics and Litter Dynamics in Mixed Deciduous Forests along an Urban-Rural Gradient, Ph.D. dissertation, Rutgers University, New Brunswick, NJ, 1992. With permission.)

Soil Temperature and Moisture

Urban environments are characterized by localized increases in temperature known as the "heat island" effect (Oke, 1990). Urban heat islands are caused by a reduction in evapotranspiration (due to reduced vegetation cover) and an increase in heat storage by urban structures. These changes result in increased minimum, and to a lesser degree increased maximum, temperatures in urban environments (Brazel et al., 2000). Temperature differences between the urban core and rural areas generally range from 5 to 10°C and usually are greatest in early evening (Brazel et al., 2000). A rise in average minimum temperature increases the number of frost-free days, effectively extending the growing season for plants, animals, and microorganisms.

The urban heat-island effect has been shown to increase soil temperatures. For undisturbed forest soils, temperatures (2-cm depth) varied by as much as 3°C across an urban-rural gradient in the New York City metropolitan area (Figure 21.4). For disturbed soils, Mount et al. (1999) found that mean annual temperatures (10-cm depth) of a playground in Central Park were more than 3°C warmer than an adjacent wooded area. In the same study, a landfill soil in Staten Island, NY, was 11°C warmer than an adjacent wooded area. The elevated temperatures in the landfill soil were attributed partially to exothermic microbial activity associated with decomposition (Mount et al., 1999).

Changes in soil temperature can have significant effects on microbial activity in soil. Pouyat et al. (1997) found that mass loss of leaf litter from litterbags was 75% greater in urban than in rural forest soils. Temperatures in the surface 2 cm of the urban forest soils were 2 to 3°C warmer than in suburban and rural soils during the field incubation period (Pouyat, 1992). Assuming a Q_{10} factor of 2, this variation in soil temperature may have accounted for 20 to 30% of the differences in decomposition rate found along the urban-rural gradient used in the study (Pouyat et al., 1997). Pouyat and Turechek (in press) found similar results using a soil transplant experiment along the same urban-rural gradient. Net N-mineralization rates of soils incubating in the urban forest stands were 46 to 62% higher than soils incubating in rural stands. The authors attributed at least some of this difference to elevated soil temperatures at the urban end of the transect.

Urban areas produce large quantities of condensation nuclei that can increase precipitation downwind of urban sources (Oke, 1990). The resultant increase in rainfall may enhance soil moisture levels. However, the potential for wetter soils can be counteracted by the tendency of soils to form hydrophobic surfaces in urban areas (Craul, 1985; White and McDonnell, 1988).

Hydrophobic soil surfaces reduce water infiltration rates, increase runoff and erosion, and lower soil water availability for microbial activity. The net result of these urban environmental effects on soil moisture regimes, and subsequently on SOC, needs to be investigated.

Atmospheric Pollution

Urban environments usually have higher concentrations and depositional fluxes of atmospheric chemicals than rural environments. Forest stands in urban areas receive relatively high amounts of heavy metals, nitrogen, and sulfur in wet and dry atmospheric deposition (Gatz, 1991; Lovett et al., 2000). High pollutant concentrations from fossil-fuel combustion, emissions from heavy industry, and temperature inversions increase deposition (Seinfeld, 1989). Consequently, urban forest soils typically have high levels of metal cations in the O and A horizons (Parker et al., 1978; Pouyat and McDonnell, 1991).

How these pollutants affect SOC pools and fluxes in urban environments is uncertain. The effects of different pollutants on decomposition are likely to be highly variable (Pouyat et al., 1997). For example, Inman and Parker (1978) found reduced rates of mass loss from litterbags in urban stands that were highly contaminated with heavy metals compared with unpolluted rural stands. However, increased rates of decomposition and N-mineralization have been reported along ozone- and N-deposition gradients associated with major metropolitan areas in southern California (Fenn and Dunn, 1989; Fenn, 1991) and New York City (Pouyat et al., 1997; Zhu and Carreiro, 1999; Pouyat and Turechek, in press) and along sulfur- and nitrogen-deposition gradients in the Midwestern United States (Kuperman, 1999).

Ozone is a plant stressor that is known to decrease forest productivity (Smith, 1990). Ozone also has the potential to affect the decomposition of leaves that are exposed prior to leaf fall. Carreiro et al. (1999) used laboratory bioassays of oak leaf litter collected along an urban-rural gradient and found that the urban-derived litter decomposed more slowly than rural-derived litter when moisture and temperature factors were held constant. The authors hypothesized that differences in leaf litter quality might be due to leaf damage by high ozone concentrations found along the urban-rural gradient. Findlay et al. (1996) reported similar results using bioassays of decaying poplar leaves exposed to high ozone concentrations.

Species Composition

In addition to urbanization-induced changes in the physical and chemical environment, nonnative plants and animals can alter the structure and composition of urban forest stands (Rudnicky and McDonnell, 1989; Zipperer and Pouyat, 1995). Nonnative species can directly or indirectly cause significant changes in ecosystem functioning, such as nutrient cycling (Vitousek et al., 1997). For example, the introduction of Asian earthworm species has altered decomposition and nitrification rates in woodlands of the New York City metropolitan area (Steinberg et al., 1997; Zhu and Carreiro, 1999). Similarly, introductions of plant species in forest stands situated in or near urban areas have been shown to alter nutrient cycling. Ehrenfeld et al. (2001) found that invasion of forests by a nonnative understory shrub and grass species resulted in soils with higher nitrification rates than soils beneath native understory species in northern New Jersey. Likewise, forest stands in the Baltimore metropolitan area dominated by nonnative tree species had higher soil nitrification rates than stands dominated by native trees (Groffman et al., unpublished data). These results suggest that nonnative species have the potential to accelerate N loss and therefore influence C fluxes in urban ecosystems.

Hydrologic Changes

Soil drainage is an important factor controlling SOC. The introduction of large areas of impervious surfaces during urban development can lead to drastic changes in soil drainage patterns, especially in wetlands and riparian soils. Decreases in infiltration due to impervious surfaces and stream-channel incision from storm-water flows can lead to decreases in water-table levels in wetlands and riparian soils (Wolman, 1967; Henshaw and Booth, 2000). In turn, changes in water-table depth can have marked effects on SOC. For example, Trettin et al. (1995) reviewed the literature and concluded that draining forested wetlands led to reductions in SOC pools. Examination of soil profiles along riparian corridors in urbanized watersheds of the Baltimore metropolitan area revealed relic drainage mottles in upper horizons of dry riparian soils (Groffman et al., in press). We hypothesize that these relic mottles were established when the water table was higher prior to the development of the watershed. Differences in current and relic mottles suggest that the drainage classes of these riparian soils were altered by at least one or two classes as water tables dropped. Similar to other studies of drained wetland soils, we suspect that this lowering of the water table has resulted in reductions in SOC. More measurements are needed, however, to document the loss of C from these urban riparian soils.

URBAN ENVIRONMENTAL CHANGES: NET EFFECTS

Given the many environmental factors that are affected by urbanization, it is difficult to predict the net effect of urbanization-induced environmental change on SOC in specific metropolitan areas. Along the urban-rural transect in New York City, Pouyat et al. (2002) found that there was no net effect of urban environmental factors on total-soil-profile SOC. However, recalcitrant C pools were higher and passive pools lower in urban compared with rural forest soils (Groffman et al., 1995). The greater proportion of recalcitrant C in the urban forest soils was attributed to inputs of poorer-quality leaf litter and enhanced mineralization of readily available C due to the activity of nonnative earthworms and increased soil temperatures (Groffman et al., 1995). Moreover, there were marked differences in the distribution of SOC within the soil profile among urban, suburban, and rural forest soils (Pouyat et al., 2002). The presence of nonnative earthworms in the urban, and to a lesser degree, suburban forest stands resulted in a decrease in the thickness of the O layer and an increase in SOC.

The New York City results show the complexity of urbanization effects on SOC. While the presence of nonnative earthworms and elevated soil temperature accelerate SOC decay, reductions in litter quality by air pollution decrease decay rates. In the absence of earthworms and with consistent annual reductions in litter quality, forest stands exposed to urban environments have the potential to sequester and store more C than rural stands (Pouyat et al., 2002). Studies of forests in other cities and the use of controlled laboratory and field experiments need to be conducted to test the generality of these results.

CHANGES IN DISTURBANCE AND MANAGEMENT REGIMES

While there are few data that address the effect of disturbance and management associated with urban development on SOC, we suggest that data from disturbance and management studies in other land-use types, e.g., conversion of native ecosystems to agriculture, should shed some light on urban land conversions. Native land uses converted to cultivation often have low levels of SOC due to physical disturbance, removal of organic material by harvest, and changes in litter quality that increase decomposition rates (Mann, 1986). The primary effect of physical disturbance is a reduction of physical protection of SOC by soil aggregates, resulting in SOC losses. Urban landscapes often have highly disturbed soils with massive or platy structure and low SOC (Craul, 1992).

In addition to a loss in structure, crop soils often are amended with fertilizer. In agricultural and grassland ecosystems, fertilization generally increases net primary production (Russell and Williams, 1982). In their review of the literature, Conant et al. (2001) found that SOC in grasslands and cultivated soils is strongly influenced by fertilization in all types of climates.

Table 21.1 Soil Organic C Densities by Land-Use Type and Proposed Classes for Disturbed and Made Soils

City/County	Type/Land Use	Proposed Classes[a]	Carbon Density (kg/m^2)
Kings, NY[b]	dredge (old)	dredgic	3.9
Kings, NY[b]	dredge (old)	dredgic	4.0
Kings, NY[b]	dredge (old)	dredgic	4.5
Kings, NY[b]	dredge (old)	dredgic	2.9
Baltimore, MD[c]	dredge (recent)	dredgic	24.7
Queens, NY[b]	refuse	garbic	13.9
Richmond, NY[b]	refuse	garbic	20.4
Moscow, Russia[d]	residential grass	scalpic	12.9
Moscow, Russia[d]	residential grass	scalpic	16.3
Chicago, IL[e]	residential grass	scalpic	18.5
Chicago, IL[e]	residential grass	scalpic	14.1
Hong Kong[f]	park use/grass	scalpic	7.2
Hong Kong[f]	park use/grass	scalpic	4.7
Hong Kong[f]	park use/grass	scalpic	3.2
Hong Kong[f]	park use/grass	spolic	3.9
Hong Kong[f]	park use/grass	spolic	2.3
Hong Kong[f]	recreational use/grass	spolic	19.1
Queens, NY[b]	recreational use/grass	spolic	28.5
Washington Monument[g]	clean fill	spolic	1.6
Richmond, NY[b]	clean fill	spolic	3.6
Richmond, NY[b]	clean fill	spolic	3.4
Richmond, NY[b]	clean fill	spolic	6.9
Washington, DC[g]	clean fill	spolic	1.4
Washington, DC[g]	clean fill	spolic	1.6
Richmond County, NY[b]	coal ash	urbic	22.9
Washington Monument[g]	construction debris	urbic	1.4

Note: Except where indicated, carbon densities were calculated with data collected from soil pit characterizations to a depth of 1 m.

[a] Class determinations based on site descriptions in Fanning and Fanning, 1989.
[b] Data from New York City Soil Survey, Natural Resource Conservation Service.
[c] Calculated from data reported in Evans et al. (2000) and data provided by Fanning.
[d] Calculated from data reported in Stroganova et al. (1998).
[e] Calculated to a depth of 60 cm from data reported in Jo and McPherson (1995).
[f] Calculated from data reported in Jim (1998).
[g] Calculated from data reported in Short et al. (1986).

A study comparing forest, cultivated, and grass (lawn) sites in the Baltimore metropolitan area found that grass areas had SOC levels similar to forests and much higher than cultivated areas (Groffman et al., unpublished data). These results are not surprising, since the grass areas have little physical disturbance and a long growing season compared with agricultural areas. However, SOC levels in grass areas likely vary with land use and management regime. Given the importance of grass cover in urban and suburban areas, there is an urgent need to understand this variation.

Pouyat et al. (2002) compiled data for pedons described and characterized for various soils in urban ecosystems. Here we add to this database and make comparisons among different soil types and land uses with different management regimes (Table 21.1). In addition, we use the anthropogenic soil classification system of Fanning and Fanning (1989), which distinguishes among several soil disturbances in urban landscapes. These include soils formed by (1) removal of a portion of the soil profile, or the scalpic soils; (2) mixing of soil material, or the spolic soils; and (3) addition of anthropogenic soil materials, or the urbic and garbic soils (Figure 21.5). Specific examples of anthropogenic soil materials (modified from Fanning and Fanning, 1989) include: for garbic soils, landfills dominated by organic waste products; spolic soils, earthy materials from industrial activities (mine spoil, dredging, highway construction); and urbic soils, materials containing more than 35%

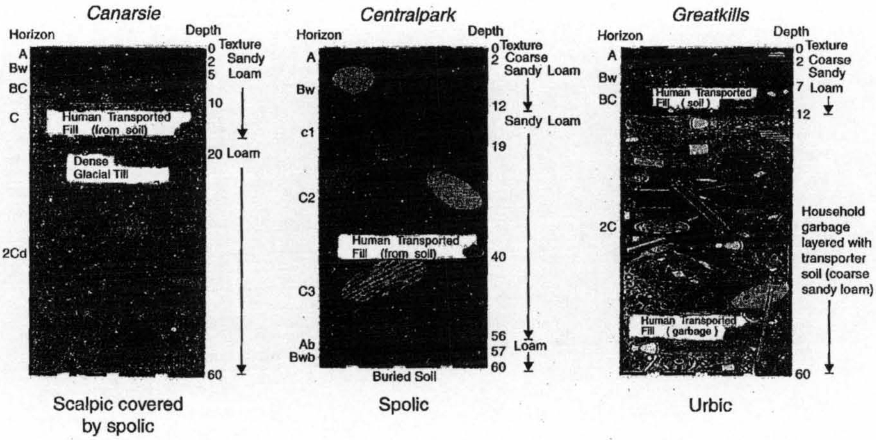

Figure 21.5 Graphical depictions of human-disturbed soil profiles including horizon designations. Profiles are (from left) examples of spolic, urbic, and scalpic classes as proposed by Fanning and Fanning (1989). The profile figures provided by New York City Soil Survey, Natural Resource Conservation Service (From Hernandez, L.A., et al., Soil Survey of South Latourette Park, Staten Island, New York, Dept. of Soil, Crop and Atmospheric Science, Cornell University, Ithaca, NY, 1997. With permission.)

(by volume) human artifacts and building rubble in the particle size control section (25 to 100 cm). We report SOC in Table 21.1 and in the following discussion as "soil C density," which accounts for the proportion of coarse fragments and bulk density of the soil on a square-meter basis to 1-m depth (Post et al., 1982).

Highly Disturbed and Made Soils

A review of the data suggests that variation in SOC density was higher among made soil types within a city than between cities for an individual soil type (Table 21.1). In New York City for example, the highest SOC density occurred on a golf course (28.5 kg/m^2), while the lowest density occurred in an old dredge site (2.9 kg/m^2), nearly a tenfold difference. For old dredge materials, however, SOC density varied by only 1 kg C m^{-2} across four different sites in two different cities. Similarly, residential yards in Chicago (Jo and McPherson, 1995) and Moscow (Stroganova et al., 1998) exhibited little variation in soil C density (15.5 ± 1.2 kg/m^2). If we compare SOC densities among the classification units proposed by Fanning and Fanning (1989), we find less consistency among SOC density values (Figure 21.6). In contrast, separating the garbic, urbic, and spolic groups by the type of fill material used results in less variation (Figure 21.6). The lack of consistency using the Fanning and Fanning (1989) groups suggests that not only the nature of the disturbance but also the origin of the soil material is an important determining factor of SOC in urban landscapes.

Managed Soils

To investigate differences in management regimes, we compare differences in SOC within land-cover types dominated by grass (Figure 21.6). In our comparison we differentiate between highly maintained recreational-use lawns to minimally maintained lawns, designations based on site descriptions associated with each pedon. Based on these assumptions, recreational-use soils had fourfold greater C densities than minimally maintained areas (Figure 21.6). We attribute these differences in SOC to differences in management intensity, e.g., fertilizer and water input. Similarly, Pouyat et al. (2002) found that the highest and most variable surface SOC densities in Baltimore

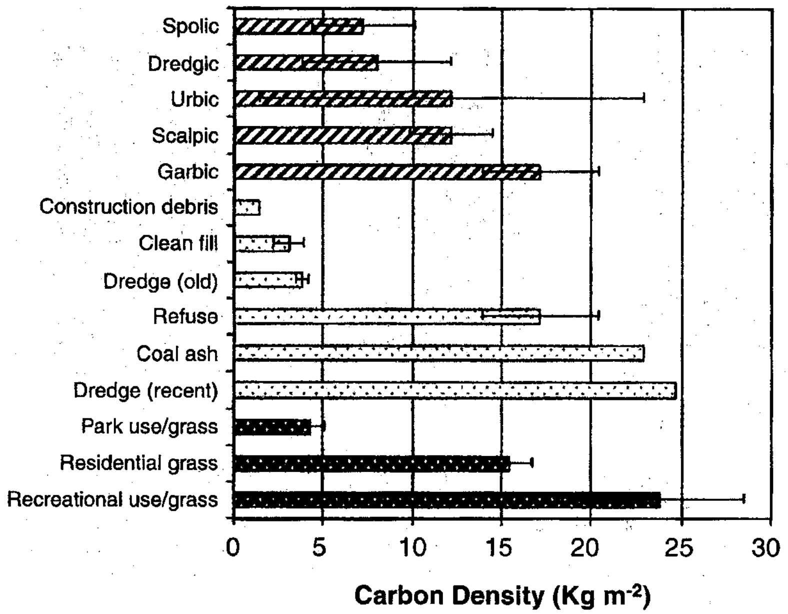

Figure 21.6 Means (±SE) of SOC densities (kg/m²) for (a) anthropogenic soil classifications proposed by Fanning and Fanning (1989); (b) the origin of material found in urbic, spolic, and dredgic soil classes; and (c) for recreational use, residential use, and park use grass cover types typically found in urban and suburban ecosystems. Data summarized from Table 21.1.

were for those land uses where lawn maintenance is expected to be relatively high, such as low-density residential and institutional land uses.

NET EFFECT OF URBAN LAND-USE CONVERSIONS

On a global scale, SOC pools are about three times larger than the C stored in all land plants (Schlesinger and Andrews, 2000). At this scale, SOC pools are primarily a function of the inputs of organic matter to the ecosystem (net primary productivity, or NPP) and the average rate of decay within the ecosystem, both of which are controlled by soil temperature and moisture. Due to differences between sensitivities of decay rate and NPP to soil temperature and moisture, a wide variation exists in SOC densities on a global scale (Table 21.2). In fact, decomposition appears to be more sensitive than NPP to variation in soil moisture and temperature (Post et al., 1982). We therefore expect the fluctuation of SOC densities to be related to SOC decay rather than to NPP (i.e., organic-matter input) as the primary effect of urban environmental changes.

Current research addresses whether SOC pools will increase or decrease with global warming (Kirschbaum, 2000) and whether various land-use changes and their associated soil modifications will affect SOC storage (Houghton et al., 1999). In the next section, we estimate changes in SOC pools at both global and regional scales to determine the net effect of urban land-use conversions.

EFFECTS AT REGIONAL AND LOCAL SCALES

Using estimates of SOC densities of disturbed and made soil types in urban areas (Table 21.1), and assuming that on average about 60% urban metropolitan land is composed of these soils, Pouyat et al. (2002) estimated that 2.63×10^{15} g and 10.7×10^{15} g of organic C exist in soils in urban

Table 21.2 Selected Life-Zone SOC Densities (at 1-m depth) and Total SOC Pools in Comparison to Urban Land on a Global Basis[a]

Life Zone[b]	Area (10^{12} m^2)	Carbon Density (kg/m^2)	Soil Carbon (10^{15} g)
Warm desert	14.0	1.4	19.6
Temperate forest — warm	8.6	7.1	61.1
Temperate thorn steppe	3.9	7.6	29.6
Urban[c, d] (total)	1.3	8.2	10.7
Tropical forest — dry	2.4	9.9	23.8
Temperate forest — cool	3.4	12.7	43.2
Temperate steppe — cool	9.0	13.3	119.7
Tropical forest — wet	4.1	19.1	78.3
Boreal forest — wet	6.9	19.3	133.2
Tundra	8.8	21.8	191.8
Wetlands	2.8	72.3	202.4
World total[e]			1500 (±20%)

[a] Modified from Pouyat et al. (2002).
[b] Data from Post, W.M. et al., Nature, 298, 156–159, 1982.
[c] World urban land total from World Resources Institute (1996).
[d] Urban land soil C density estimate based on data presented in Table 21.1. Urban SOC density data biased toward warm, temperate life zones.
[e] World total estimate from Schlesinger and Andrews (2000).

ecosystems on a national and global basis, respectively (Table 21.2). These estimates are based on aerial coverage of urban land use calculated by Nowak et al. (1996) and the approximate coverage of urban land (3.5%) for the contiguous 48 United States (Dwyer et al., 1998). The authors' estimate of SOC storage in urban ecosystems does not represent a significant proportion of the national total for SOC storage (Table 21.2). Likewise, on a global scale, urban areas make up approximately 1% of the land base and only 0.7% of the SOC pool (Table 21.2).

A comparison of forested areas with residential areas, which make up a large proportion of urban land, suggests that residential areas have nearly the same C density as northeastern forests (Figure 21.7). As mentioned previously, the relatively high SOC densities in residential areas are due most likely to increases in ecosystem productivity from fertilizer and water applications. Lawns also have a much longer growing season than forests. On a global scale, residential soils are more similar to cool temperate steppe life zones than to Northeast forest soils in SOC density (Table 21.2 and Figure 21.7), a result that is consistent with observations of plant community structure in urban ecosystems (e.g., Dorney et al., 1984).

While residential soils are similar to Northeastern forests and steppe ecosystems, disturbed and "made" soils have similar C densities to Northeastern and Mid-Atlantic croplands (Figure 21.7). Both cropland and lawns may be treated with fertilizer or irrigated. The main difference between these cover types, however, is the physical disturbance resulting from cultivating cropland soils, which eventually leads to a reduction in SOC (Conant et al., 2001). For various land use and cover types sampled in the City of Baltimore, physical disturbances resulting in soil compaction (as measured by bulk density) were found to explain almost 40% of the variation in SOC pools in the surface 15 cm of soil (Pouyat et al., 2002).

Carbon storage will increase or decrease depending on the region of the country as urban and suburban areas expand. For the soils included in Table 21.1, Pouyat et al. (2002) estimated an average C density of 8.2 kg m^{-2}, which for the northeastern United States would represent a decline in SOC storage relative to native soils prior to urbanization (Figure 21.7). For the Mid-Atlantic states, however, this C storage estimate represents an increase (Figure 21.7). It is unclear what the net result would be in other regions of the country as this estimate of urban SOC density utilizes data mainly collected for cool to warm temperate areas.

Estimates of SOC and comparisons among ecosystems and cover types are preliminary and may vary considerably among cities within the United States and globally. Furthermore, consider-

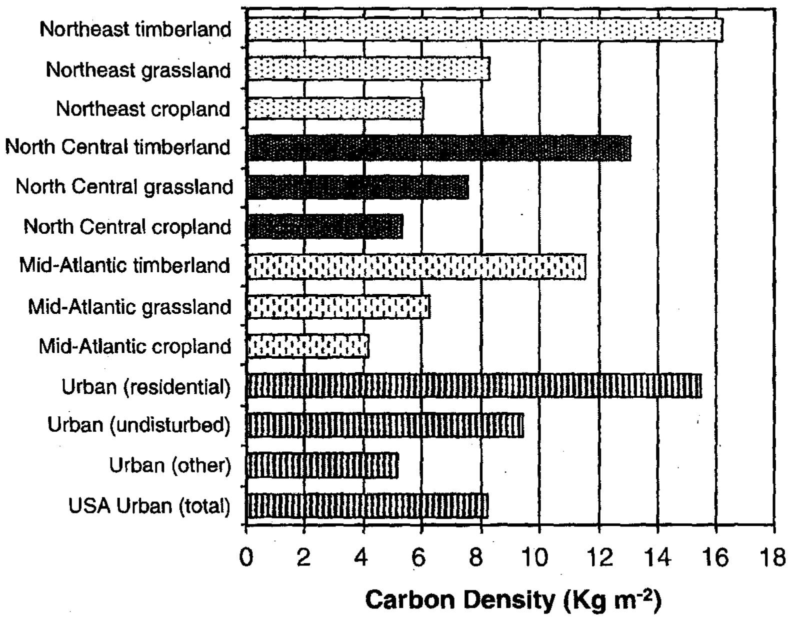

Figure 21.7 Comparison of SOC densities (kg/m² at 1-m depth) for disturbed, managed, and undisturbed soils found in urban ecosystems with forest and cropland soil estimates for the Northeast, North Central, and Mid-Atlantic states. Urban soil types were compiled from soil pedon data presented In Table 21.1. Carbon densities of forest and cropland soils from Birdsey (1992). Aerial coverages of forest and cropland soils calculated from Table 21.1 (Modified from the USDA Forest Service, 1997).

able variation can occur among soils in urban landscapes (Figures 21.3 and 21.6), and therefore the C density estimate in Table 21.1 should not be considered representative of all soils found in urban areas.

URBAN ECOSYSTEM CONVERGENCE HYPOTHESIS

As a database is developed for soils situated in urban landscapes, it would be interesting to compare soils in cities located in various climates with those included in Table 21.1. We suggest an "urban ecosystem convergence hypothesis," where urbanization drives ecosystem structure and function (e.g., SOC pools, leaf area index), toward a range of similar endpoints over time regardless of ecosystem life zone starting points (Figure 21.8).

We propose the following arguments to support our hypothesis. First, urban ecosystems are novel habitats that tend to be dominated by similar assemblages of species. We propose that this consistency in plant and animal community composition across urban ecosystems is due to similarities in urban environments across regional and global scales and to the tendency of humans to disperse many of these species (intentionally or nonintentionally). Second, humans adopt land-management practices that attempt to overcome site limitations. For agricultural purposes, poorly drained soils are drained while excessively drained soils are irrigated to provide optimum conditions for plant growth. Similarly, urban and suburban turf-management practices attempt to maximize the health of turf grasses or ornamental plants by amending soils.

There is evidence for a "convergence" of SOC pools from comparisons made of agricultural land conversions on regional and global scales. Post and Mann (1990) found that the amount of SOC lost after cultivation was a function of the initial amount of SOC stored in soil. Specifically, the average loss for soils with high initial SOC was about 23%, while soils with low initial SOC

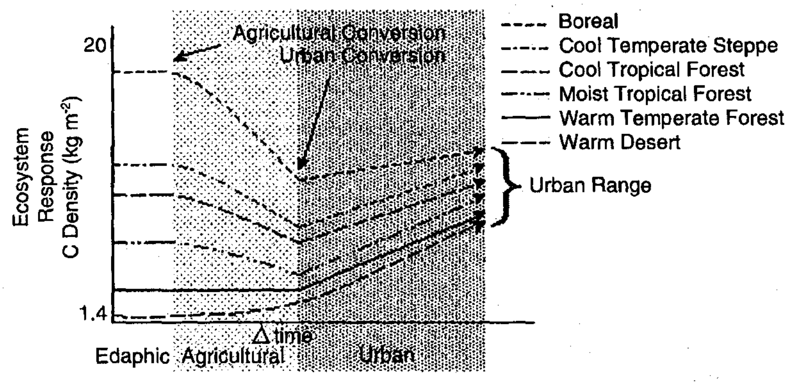

Figure 21.8 Plot of hypothesized ecosystem response to agricultural and urban land-use conversions over time. In this example, soil carbon density (kg/m^2) for boreal, cool temperate steppe, cool tropical forest, moist tropical forest, warm temperate forest, and warm desert life zones (Table 21.1) converge on a range of soil carbon densities following urbanization.

actually increased their C storage during cultivation. Our preliminary comparisons of changes in SOC pools in post-urban development between the northeastern and Mid-Atlantic states showed similar trends (Figure 21.7).

Obviously, the amount of change in SOC pools due to urban development will depend on the extent of land that is managed and disturbed by humans. This will vary by the age of a city, socio-economic factors, and the predevelopment geomorphology of the area. We believe, however, that on regional and global scales, the change in SOC will be similar to that observed in agricultural systems — the direction of change will depend on the initial SOC status of the native soil (Figure 21.7). A significant outcome of this hypothesized relationship is that it cannot be assumed that urbanization will result in a net loss or net gain of SOC from soil.

FUTURE RESEARCH NEEDS

Our review of the literature suggests that more data are needed to determine the net effect of human modification on SOC pools and thus make accurate estimates of SOC in urban ecosystems at regional and global scales. Urban ecosystems are composed of highly disturbed soils, soils covered by impervious surfaces, made soils, highly managed soils, and relatively natural soils. These soil conditions need to be investigated to determine initial SOC pools (Table 21.2). Uncertainties include the quality of the C inputs, the fate of SOC soils covered by impervious surfaces, measurements of SOC at depths greater than 1 m, and spatially delineating disturbed and "made" soil types. Data are needed to assess the differences in quality of nonnative and native plant species litter and the effect of stress on litter quality. Data also are needed for highly disturbed soils, particularly "made" soils (e.g., landfill), at depths greater than what is normally characterized by soil scientists (i.e., greater than 2 m). Effects of cultivation on SOC pools are largely understood. But little is known about other soil disturbances, such as compaction and massive soil movement due to cut-and-fill operations. Measurement of SOC pools represents a challenge, since little data exists on soil bulk density, percentages of rock fragments, and human artifacts in made soils.

In addition to measuring SOC pools in various soil types, investigations also are needed to determine the sensitivity of SOC decay to various urban environmental changes and management regimes. Moreover, more investigations are needed in urban and suburban ecosystems to measure inputs of plant residues to the soil decomposer system. These types of measurements are needed to determine the long-term effects of both soil physical and environmental changes and management regimes on soil C fluxes. Urbanization also can have major effects on riparian and wetland ecosystems, which can lead to the release of large amounts of C to the atmosphere.

CONCLUSION

Based on our review of the available data, SOC pools can be affected directly and indirectly by urbanization; the direction of change depends on the initial amount of SOC in the native soil. Indirect effects of urban environments on soil C have been shown to be complex and variable. In oak stands along an urban-rural transect in the New York City metropolitan area, nonnative earthworms, differences in soil temperature, and changes in litter quality were important. It is not clear if these factors can be generalized across urban ecosystems. However, results from these New York City studies suggest that urban environmental changes can affect soil C pools even in forests that are not directly or physically disturbed by urban development.

Our analysis also suggests that soil C storage in urban ecosystems is highly variable, with high and low SOC densities (kg/m^2 to a 1-m depth) present in the landscape at any one time. For those soils with low SOC densities, there is potential to increase C sequestration through management, but specific urban-related management techniques need to be evaluated. Though soils in urban landscapes store relatively small absolute amounts of C, it is important to note that changes associated with urbanization likely are to be more persistent than changes associated with other land-use conversions. Moreover, with the growing importance of land-use conversion from native to urban and suburban ecosystems, it is important to consider these effects when calculating C budgets for localities and regions experiencing rapid urban expansion.

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