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Environmental Pollution 116 (2002) S107 S118

ENVIRONMENTAL
POLLUTION

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Soil carbon pools and fluxes in urban ecosystems

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"Capsule": *Soil organic carbon pools are directly and indirectly affected by urban land-use conversions and these changes can be observed in adjacent undisturbed forests.*

Abstract

The transformation of landscapes from non-urban to urban land use has the potential to greatly modify soil carbon (C) pools and fluxes. For urban ecosystems, very little data exists to assess whether urbanization leads to an increase or decrease in soil C pools. We analyzed three data sets to assess the potential for urbanization to affect soil organic C. These included surface (0–10 cm) soil C data from unmanaged forests along an urban rural gradient, data from "made" soils (1 m depth) from five different cities, and surface (0–15 cm) soil data of several land-use types in the city of Baltimore. Along the urban–rural land-use gradient, we found that soil organic matter concentration in the surface 10 cm varied significantly ($P=0.001$). In an analysis of variance, the urban forest stands had significantly ($P=0.02$) higher organic C densities (kg m^{-2} to 1 m depth) than the suburban and rural stands. Our analysis of pedon data from five cities showed that the highest soil organic C densities occurred in loamy fill (28.5 kg m^{-2}) with the lowest occurring in clean fill and old dredge materials (1.4 and 6.9 kg m^{-2}, respectively). Soil organic C densities for residential areas (15.5 ± 1.2 kg m^{-2}) were consistent across cities. A comparison of land-use types showed that low density residential and institutional land-uses had 44 and 38% higher organic C densities than the commercial land-use type, respectively. Our analysis shows that as adjacent land-use becomes more urbanized, forest soil C pools can be affected even in stands not directly disturbed by urban land development. Data from several "made" soils suggests that physical disturbances and inputs of various materials by humans can greatly alter the amount C stored in these soils.

Keywords: Soil organic carbon; Anthropogenic soils; Urban soils; Human modified soils; Baltimore Ecosystem Study

1. Introduction

The expansion of urban areas worldwide makes our understanding of the effects of urbanization on soils increasingly important. Over 50% of the world's population lives in urban areas (World Resources Institute, 1996). In the lower 48 states of the USA alone, urban areas have increased two-fold in area between 1969 and 1994, and currently occupy 3.5% of the land base (Dwyer et al., 1998). On a global scale over 476,000 ha of arable land is annually being lost to the expansion of urban areas (World Resources Institute, 1996).

The transformation of landscapes from primarily agricultural and forest uses to urbanized landscapes has the potential to greatly modify soil carbon (C) pools and fluxes (Groffman et al., 1995; Pouyat et al., 1995b). While conversion of native ecosystems to agricultural use, and recovery from agricultural use, have been relatively well-studied, conversions to urban land uses have received little attention. We know, for example, that as agricultural practices have been abandoned in previously forested areas in the United States, regrowth in these soils has resulted in a gradual recovery of above ground C pools (Houghton et al., 1999; Caspersen et al., 2000). Urban land conversions, however, often result in poor conditions for plant growth, and if the land is abandoned (an unlikely occurrence for large geographical areas) recovery should be slower than on previously agricultural lands (e.g. Clemens et al., 1984).

On a global scale soil C pools are roughly three times larger than the C stored in all land plants (Schlesinger and Andrews, 2000). At this scale soil C 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 (soil heterotrophic respiration), both of which are controlled by environmental factors such as soil temperature and moisture. Due to differences between sensitivities of decay rate and NPP to soil temperature and moisture, a

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PII: S0269-7491(01)00263-9

wide variation exists in organic soil C densities (kg m^{-2} to 1 m depth) on a global scale (Post et al., 1982). Current research efforts are addressing the question of whether soil C pools will increase or decrease when global warming occurs (Kirschbaum, 2000) and whether various land-use changes and their associated soil modifications will affect soil C storage (Houghton et al., 1999). The long-term effects of forest harvesting, for example, is negligible when regrowth of the forest occurs (Johnson, 1992). Agricultural uses, however, have generally led to greater losses of soil organic C (Paul and Clark, 1996). For urban ecosystems, very little data are available to assess whether urbanization leads to an increase or decrease in soil C pools. This paucity of data has made it problematic to predict or assess the regional effects of land use change on soil C pools in various regions of the world (e.g. Howard et al., 1995; Ames and Lavkulich, 1999).

As land is converted to urban uses both direct and indirect factors can affect soil C 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). Pouyat and Effland (1999) suggest that direct effects often lead to "new" soil parent material on which soil development then proceeds. Indirect effects involve changes in the abiotic and biotic environment as areas are urbanized that can influence soil development in intact soils. Indirect effects include, the urban heat island effect (Oke, 1995; Mount et al., 1999), soil hydrophobicity (White and McDonnell, 1988; Craul, 1992), introductions of exotic plant and animal species (Airola and Bucholz, 1984; Steinberg et al., 1997), and atmospheric deposition of various pollutants (Grodzinski et al., 1984; Pouyat and McDonnell, 1991; Lovett et al., 2000). Moreover, toxic, sub-lethal, or stress effects of the urban environment on soil decomposers and primary producers can significantly affect soil C fluxes (Pouyat et al., 1994, 1997; Goldman et al., 1995; Groffman et al., 1995; Carreiro et al., 1999).

In this paper we discuss the potential for soil disturbances and various urban environmental changes to affect soil C pools and fluxes in urban ecosystems. Specifically we address the following questions: (1) How large are existing soil organic C pools in urban ecosystems and how do they compare to the native ecosystems replaced by urbanization? and (2) How do soil organic carbon pools vary across different land-use types in urban landscapes? To address these questions and explore the importance of urban indirect effects (environmental factors) on soil C pools, we present a case study conducted in the New York City metropolitan area where the structure and function of unmanaged oak forests were compared across an urban–rural environmental gradient (McDonnell et al., 1993). To show how direct effects, or soil disturbances, can modify soil C pools, we present data from newly described soil pedons (largely human "made" soils) collected as part of the New York City soil survey and other data mined from the literature. Finally, to explore at a city-wide scale the spatial variation in soil C pools resulting from these urban effects, we present preliminary data from 127 plots in the city of Baltimore.

2. Study areas

2.1. New York City urban–rural gradient

We have been investigating forest soils along an urban-rural land-use gradient in the New York City metropolitan area since 1990 (Groffman et al., 1995; Pouyat et al., 1995a, b; McDonnell et al., 1997). This research includes detailed characterization of the gradient including the quantification of land-use characteristics, soil chemical and physical properties, soil C and N dynamics, and ecological analysis of plant community structure and soil organism abundances.

The study area includes contrasting land uses extending from Bronx County, New York (New York City) to sites in Westchester County, New York, and Litchfield County, Connecticut (McDonnell et al., 1993). The climate of the region is characterized by warm humid summers and cold winters. Average annual air temperatures range from 12.5 °C in New York City to 8.5 °C in northwestern Connecticut (NOAA, 1985). Much of this difference in air temperature is attributed to the heat island effect associated with New York City (Bornstein, 1968). Precipitation is distributed evenly throughout the year for the entire study area and ranges from an annual average of 108 cm in New York City to 103 cm in northwestern Connecticut (NOAA, 1985).

The Bronx, New York, and the study area to the north constitute the southern portion of the Northeastern Upland Physiographic Province (Broughton et al., 1966). The bedrock consists of highly metamorphosed and dissected crystalline rocks that are composed of schist, granite and gneiss (Schuberth, 1968). The soil types included in the study are well-drained, moderate to shallow, sandy loam soils situated on gently sloping terrain (Hill et al., 1980). A more detailed description of the study area is given in Medley et al. (1995) and Pouyat (1992).

In a previous study, a 20 km wide by 130 km long belt transect was established along an urban–rural land-use gradient in the study area (Pouyat and McDonnell, 1991). The transect encompasses readily measurable differences in population density, road density, and automobile usage and corresponds closely to the location of gneiss and schist bedrock in the study area (Pouyat et al., 1995a). Forest stands were selected for the study using the following criteria: (1) location on upland sites on well-drained, moderate to shallow, sandy loam Inceptisols in the Dystrochrepts great group that vary only in depth to bedrock (Gonick et al., 1970; Hill et al., 1980); (2) oak-dominated forest with *Quercus rubra* L. and *Quercus velutina* Lam., both of the subgenus *Erythrobalanus*, as major components of the overstory (40–80% of total basal area); (3) minimum stand age of 70 years; and (4) no visual evidence of natural disturbance, such as fire or logging.

By design, none of the study sites included a significant proportion of non-native tree species. Measurements in previous studies, however, showed differences in abundances of non-native species of earthworms and soil organic horizon characteristics between plots. The urban, suburban, and rural plots had on average 25.0, 7.0, and 2.5 individuals m^{-2} of non-native species of earthworms, respectively (Steinberg et al., 1997). Moreover, all of the urban plots and a few suburban plots have mull soil organic horizons (Pouyat, 1992; Zhu and Carreiro, 1999), which are characteristic of forest soils with high earthworm densities. In addition to variations in earthworm density (no. of individuals m^{-2}), 2–3 °C higher temperatures and up to 5-fold differences in heavy metal and total salt concentrations were measured in the upper 10 cm of soil in the urban than in the suburban and rural stands (Pouyat et al., 1995a). Soil physical properties, such as bulk density and texture, did not differ appreciably among stands (Table 1). Data on organic matter and C concentrations in these soils have been reported in previous studies (Groffman et al., 1995; Pouyat et al., 1995a). In this paper we analyze surface mineral soil and forest floor organic C pools from the same stands.

Table 1
Means (±S.E.) of soil chemical and physical properties (10 cm) by land-use type along an urban–rural transect in the New York City metropolitan area, spring 1989[a]

Soil properties	Means by land-use		
	Rural	Suburban	Urban
PH	4.7	4.6	4.5
Conductivity (S m^{-1})	0.081 (0.002)	0.115 (0.003)	0.131 (0.004)
Bulk density (Mg m^{-3})	0.88 (0.02)	0.91 (0.02)	0.87 (0.02)
Sand (%)	75 (0.61)	74 (0.94)	74 (0.63)
Clay (%)	9.2 (0.37)	10.1 (0.47)	9.6 (0.34)
Cu (mg kg^{-1})	14.8 (1.94)	16.0 (0.68)	31.6 (1.14)
Ni (mg kg^{-1})	14.0 (0.60)	17.8 (1.06)	22.0 (1.25)
Pb (mg kg^{-1})	26.9 (0.86)	36.7 (2.06)	110.0 (4.12)
Ca (mg kg^{-1})	28.3 (6.2)	61.2 (9.8)	175.6 (23.7)
Mg (mg kg^{-1})	11.3 (0.69)	13.4 (0.76)	29.8 (2.8)
K (mg kg^{-1})	48.2 (2.7)	54.5 (2.7)	65.3 (3.9)
N (g kg^{-1})	1.97 (0.14)	2.31 (0.11)	2.63 (0.11)
Organic matter (g kg^{-1})	73 (4.3)	83 (2.6)	97 (3.3)

[a] Values are the mean of nine plots (four composite samples per plot) for each land-use type (adapted from Pouyat, 1992).

2.2. New York City soil survey

New York City is approximately 80 500 ha in area and is mainly comprised of a group of islands, politically grouped into counties or boroughs. These include Kings (Brooklyn) and Queens Counties that are both located on the western portion of Long Island, and New York (Manhattan), Bronx, and Richmond (Staten Island) Counties. Richmond County is separated from the other counties by the Hudson River.

Surficial geology consists of glacial drift (Pleistocene epoch) and post-glacial deposits (Schuberth, 1968). The Harbor Hill terminal moraine extends across the southern part of the city crossing Queens, Brooklyn, and Staten Island. Glacial drift in Staten Island is redder in color (10YR or redder using Munsell system) than the drift in Queens and Brooklyn. Two glacial lakes formed on the back side of the terminal moraine submerging the island of Manhattan, the western portion of Staten Island and northern portion of Queens and Brooklyn. Outwash plains formed on the front side of the terminal moraine in Queens, Brooklyn and Staten Island.

Structural geology of New York City is very complex and consists of eight bedrock formations (Schuberth, 1968). These range from gneiss and schist formations in the Bronx to diabase and serpentine dominated formations in Staten Island. Post-glacial formations consist of organic, eolian and anthropogenic materials. Eolian materials mostly occur on barrier islands along the Atlantic coast. These Barrier islands formed lagoons where organic deposits have accumulated during the past 4000 years.

2.3. Baltimore City plots

The Baltimore metropolitan area has hot humid summers and cold winters with average annual air temperatures ranging from 14.5 °C in Baltimore City to 12.8 °C in the surrounding area (NRCS, 1998). This difference in air temperature between the city and surrounding areas is attributed to the heat island effect associated with Baltimore City (Brazel et al., 2000). Precipitation is distributed relatively evenly throughout the year for the entire study area and ranges from an annual average of 107.5 cm in Baltimore City to 104 cm in the surrounding metropolitan area (NRCS, 1998).

Baltimore City lies within two physiographic provinces, the Piedmont Plateau and the Atlantic Coastal Plain. The north–northeast trending Fall Line separates the two provinces, dividing the city in half. Most of the city is characterized by nearly level to gently rolling uplands, dissected by narrow stream valleys. Old igneous and metamorphic rocks underlie the Piedmont Plateau in the City of Baltimore. Much younger, poorly consolidated sediments underlie the Coastal Plain in the city. Four soil Associations make up most of the City's area. These include: (1) Urban Land-Legore Association, which are very deep, nearly level to moderately sloping, well drained upland soils that are underlain by semibasic or mixed basic and acidic rocks (34% of the land area, of which 43% is urban land); (2) Urban Land-Joppa-Sassafras Association, which are very deep, somewhat excessively drained and well-drained upland soils that are underlain by sandy or gravelly sediments (18% of the land area, of which 40% is urban land); (3) Urban Land-Sunnyside Association, which are very deep, nearly level to moderately sloping, well-drained upland soils that are underlain by unstable clayey sediment (24% of land area, of which 54% is urban land); and (4) highly disturbed soils that vary in slope position, drainage, and origin that make up 24% of the land area (NRCS, 1998). Based on the percentage of cover of these soil associations, the total land cover of the city that is made up of highly disturbed soils (urban land and Udorthents) is approximately 60%, though the coverage of undisturbed soil map units by impervious surfaces, such as roads, is not included in this estimation.

3. Materials and methods

3.1. New York City urban–rural gradient

Twenty-seven plots were established along the urban–rural land-use gradient. Each plot was 20×20 m and consisted of sixteen 5×5-m (0.025 ha) quadrats, four of which were randomly selected for sampling. Mineral soil, or soil below the O horizon, was sampled to a depth of 10 cm using a standard 2-cm diameter stainless steel sampling probe. Approximately 10 cores were composited for each quadrat. Two 5×5-cm cores were taken per plot to determine bulk density. Mineral soil samples were air dried, ground and sieved. Organic matter concentration of mineral soil was determined by loss on ignition (450 °C for 4 h). The entire organic soil layer was sampled together to a depth that reached the mineral soil surface, using a 15×15 cm (225 cm^2) template for each quadrat. Organic layer samples were separated into O_1 (corresponding to O_i horizon) and O_2 (corresponding to O_a and O_e horizons) layers, oven dried at 66 °C to constant weight, weighed and ground in a Wiley Mill with a 20-mesh stainless steel screen. Subsamples of soil and forest floor were analyzed for total C and N using a Carlo Erba NA1500 Analyzer. Carbon amounts (kg C m^{-2}) in the forest floor O-layers were calculated from concentration (g kg^{-1} dry mass) and dry mass (kg dry mass m^{-2}) data. Organic C densities (kg C m^{-2}) in the surface mineral soil (0–10 cm depth) were calculated by multiplying concentration (mg C g^{-1} soil) by the areal density (g soil cm^{-2}; Garten et al., 1999). The aerial density of the soil was calculated as the product of the bulk density (g soil cm^{-3}) and the depth of the sample (10 cm). Coarse fractions were estimated using pedon data from the New York City Soil Survey Database, which ranged between 2 and 3% in the mineral surface horizons for the soils in this study. The mineral soil organic C pools are expressed on a kg m^{-2} basis. Mineral soil organic C amounts were added to forest floor amounts to derive the forest soil organic pool (O layers + surface 10 cm mineral soil).

Urban environmental effects were evaluated statistically in two ways. First, soil organic matter and C concentrations, and mineral and forest soil (mineral + O_1 + O_2) organic C densities were regressed against distance to the urban core. Second, data were combined into urban, suburban, and rural land-use classes (nine plots per each land use) and subjected to a one-way analysis of variance (ANOVA) to test for differences in organic C amounts between land-use types.

3.2. New York City soil survey

The New York City soil survey is being conducted at two scales: a reconnaissance soil survey of the entire city and high intensity soil surveys of specified areas (Hernandez and Galbraith, 1997; Hernandez, 1999). The reconnaissance survey was conducted as a medium intensity soil survey (order 3) utilizing the standard classification system, but with these added features: a higher level of analysis (at the series level), mapping at a smaller scale (1:62,500), and use of proposed new soil series. The intensive soil surveys (order 1) were conducted using large mapping scales (1:4800 and 1:6000). Large-scale surveys have been completed at South Latourette Park in Staten Island (Hernandez and Galbraith, 1997) and are currently underway at Gateway National Recreation Area.

The United States National Cooperative Soil Survey (NCSS) Program has prepared soil maps for much of the country, however, this effort has focused primarily on agricultural areas. Mapping and classification of urban areas requires the establishment of new soil series. In the New York City soil survey, 35 new soil series have been established for human disturbed soils. Only 10 of the 35 pedons, however, have been completely described and were used in this analysis. Soil series names of highly disturbed soils have been established by defining a specific range of characteristics. Soil features that have been used to develop soil series concepts include the type of transported material, anthropogeomorphic processes, thickness of fill material, bulk density, organic carbon distribution, base saturation, trace elements, soil temperature, presence of a diagnostic horizon after disturbance, and amount and kind of human artifacts (Hernandez, 1999).

The following sampling procedures have been developed for the New York City soil survey. More specific sampling procedures and laboratory analysis methods can be found in the Soil Survey Manual (Soil Survey Staff, 1993) and Soil Survey Laboratory Methods Manual (Soil Survey Staff, 1996). The soil series type location was carefully selected using transect observations, landscape models, and Major Land Resource Area (MLRA) soil survey documentation. The selected pedon for sampling represents the central concept of the series. After selecting the site, a 1.5 m^2 soil pit was excavated with a backhoe or by hand, to either a depth of 1.8 m or contact to bedrock, whichever is shallower. Soil horizons, or zones of uniform morphological characteristics, were identified and were described using NRCS guidelines for describing and sampling soils (Schoeneberger et al., 1998). A representative sample from each horizon was collected. A subsample was used to measure organic C concentrations using $FeSO_4$ Titration (method 6A1c; Soil Survey Staff, 1996). For those cases where organic C data were not available, organic C concentration was calculated by multiplying organic matter concentration by the Van Bemmelen factor, or 0.58 (Soil Survey Staff, 1992). In the field, the 20- to 75-mm fraction was sieved, weighed, and discarded. Samples taken to the laboratory were sieved and weighed to determine the <20-mm fraction. Weight percentages of the >2-mm fractions were estimated from volume estimates of the >20-mm fractions and weight determinations of the <20-mm fractions by procedure 3B1b (Soil Survey Staff, 1996).

Undisturbed clods were collected for bulk density and micromorphological analysis. Clods were obtained in the same pit face as the mixed, representative sample. Four bulk density clods were collected from each horizon. Two of the clods were used in the primary analysis, while the third clod was reserved for a rerun, if needed. The fourth clod was collected for preparation of thin sections and micromorphological examination (data not included in our analysis).

We calculated the density of C in a horizon of unit area (1 m^2) as:

$$c = c_f B_D (1 - \delta_{2mm}) V$$

where c is carbon density, δ_{2mm} is the fraction of material larger than 2 mm diameter, B_D is bulk density, c_f is the fraction by mass of organic C, and V is the volume of the horizon (Post et al., 1982). Data for the soil horizons were summarized to report soil C density on a m^2 basis to a 1 m depth.

For this paper, pedon data from disturbed soils were assigned into "made" and residential soil categories. Pedon data of disturbed soil profiles in urban areas were located in the literature (Short et al., 1986; Jo and McPherson, 1995; Stroganova et al., 1998; Evans et al., 2000) and added to the New York City data set. The made soils categories were further subdivided based on the origin of the fill material (loamy fill, clean fill, refuse, old dredge, and recent dredge materials).

3.3. *Baltimore City plots*

Plots were located in Baltimore City by a stratified random design. Seven land-use types were delineated using 1994 Digital Orthophoto Quarter Quads (Maryland Department of Natural Resources) and were weighted based on the aerial coverage of each type. These land-use types included commercial; industrial; institutional; transportation right-of-ways; high, medium, and low density residential; and forest. A grid was laid over the land-use map and 200 plots were randomly located on the grid.

At each sample point a circular plot with an 11.35 m radius was established. In a previous study, plant species, vegetation structure, and other measurements were recorded in each plot (D. Nowak, unpublished data). During the summer of 2000, 127 of the original 200 plots were sampled for soils. Fewer plots were sampled because many sample points landed on impervious surfaces, or permission was not granted to collect soil samples at a number of private residences. If more than one type of cover occurred within a plot, samples were stratified by the cover types present. Cover types included tree, managed grass, unmanaged herbaceous, and "no vegetation" categories. Two techniques were used to acquire soil samples within the plots: an undisturbed 5-cm bulk density core (three per cover type) and a composite soil sample to a depth of 15 cm with a 2-cm diameter stainless steel sampling probe. Typically, 10–15 cores (2 cm) were sampled in a grid pattern and composited, depending on the amount of soil surface that was exposed. The composite sample was air-dried and sieved in the lab with a 2-mm mesh sieve. The undisturbed cores were used to determine soil bulk density. The dry sieved samples were used to measure various soil properties, of which organic matter content will be reported here. Organic matter concentration of mineral soil was determined by loss on ignition (450 °C for 4 h). Soil organic C was estimated by multiplying organic matter content by 0.58 (Soil Survey Staff, 1992). Organic C amounts (kg C m^{-2}) in the mineral soil were calculated the same as for the urban-rural gradient study described above except the depth used was 15 cm and the fraction of coarse fragments (>2 mm) were not factored into these calculations. For this analysis, we present soil organic C data for each of the land use categories used to stratify the sampling.

4. Results

4.1. *New York City urban–rural gradient*

Forest floor mass and soil organic C varied widely among oak forests along the urban–rural transect. A significant ($P = 0.01$) relationship existed between soil organic matter concentration (g kg^{-1}) in the surface 10 cm and distance from the urban core with approximately 23% of the variation among plots explained by distance. Unlike soil organic matter, both organic C density and concentration in the surface 10 cm of soil were not statistically significant when regressed against distance ($P=0.07$ and 0.06, respectively). The trend for all three measurements was to decrease from the urban to the rural end of the transect with negative slopes (−0.22, −0.13, and −0.01 for soil organic matter concentration, C concentration, and C density, respectively).

Regressions of total forest floor, O_1, and O_2 layer mass against distance to the urban core were not statistically significant. However, when plots exhibiting mull humus layers were excluded from the regression the relationship between total forest floor mass and distance was statistically significant ($P=0.0004$; Fig. 1). The regression between distance and soil organic matter is similar to the regression with forest floor mass (absent mull humus plots), where mass decreased from the urban to the rural end of the transect with a negative slope (−0.03). Distance from the urban core explained 54% of the variation of forest floor mass (absent mull humus plots) among plots (Fig. 1).

ANOVA-based comparisons of organic C densities (kg C m^{-2}) of urban, suburban, rural land use types showed contrasting trends for different horizons. Mineral soil organic C densities were approximately 30% higher ($P=0.03$) in urban compared to suburban and rural stands (Fig. 2). Forest floor organic C densities showed an opposite, but statistically insignificant trend ($P=0.43$) as the mineral soil densities (Fig. 2). The result of these contrasting responses was that differences among the land-use types in forest soil organic C densities (soil + forest floor organic C) were not statistically significant ($P=0.16$).

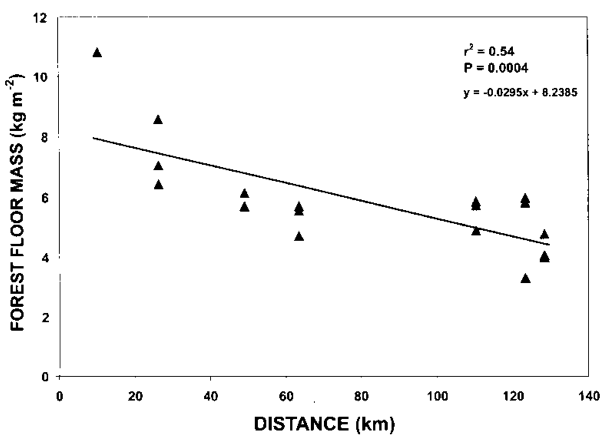

Fig. 1. Plot of forest floor mass as a function of distance to the urban core for data collected along urban-rural transect in New York City metropolitan area. Values are the mean of nine forest plots (four composite samples per plot).

To explore the importance of earthworm activity on surface soil organic C pools, we compared C pools between mull and mor soil humus types. Mineral soil organic C densities were approximately 39% higher ($P=0.02$) in mull than in mor humus types (Fig. 3). In contrast, forest floor organic C densities were more than 2-fold higher ($P=0.001$) in the mor than in the mull humus types (Fig. 3). No differences occurred in forest soil (mineral + O layer) organic C between mor and mull humus types, suggesting that while earthworms influence distribution of organic matter in the soil profile, they do not affect total forest soil C content (Fig. 3).

4.2. New York City soil survey

Soil organic C densities to a 1 m depth varied widely across disturbed or "made" soil types. Variation in soil

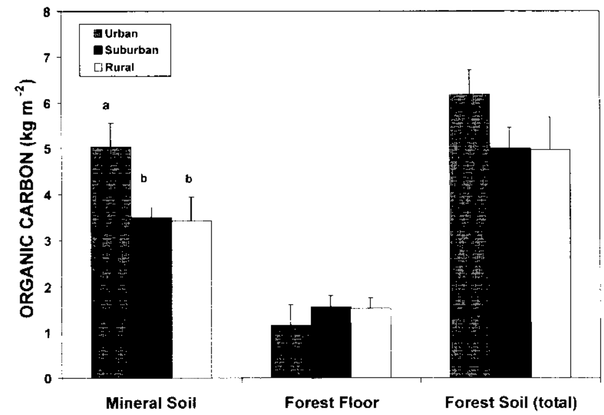

Fig. 2. Means (±S.E.) of organic C densities (kg m^{-2}) for urban, suburban, and rural forest stands. Bars are grouped by mineral soil (0–10 cm), forest floor (O_1+O_2), and total forest soil (forest floor + mineral soil). Values are the means of nine forest plots (four composite samples per plot).

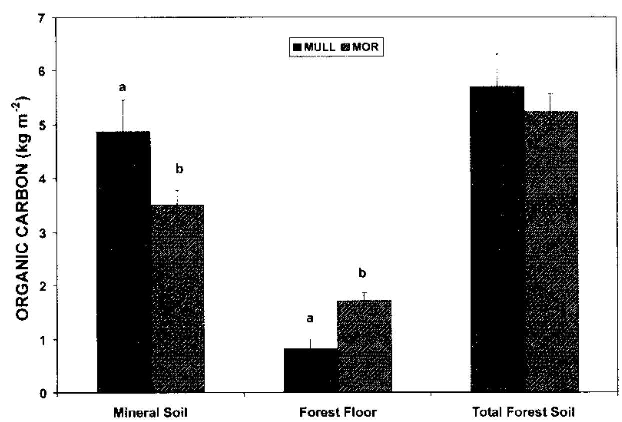

Fig. 3. Means (±S.E.) of organic C densities (kg m^{-2}) for mull and mor humus types. Bars are grouped by mineral soil (0–10 cm), forest floor (O_1+O_2), and total forest soil (forest floor + mineral soil). Values for mull and mor humus types are the means of nine and 18 forest plots, respectively (four composite samples per plot).

Table 2
Soil organic C densities for disturbed and made soils in the cities of New York, NY, Chicago, IL, and Moscow, Russia (except where indicated, carbon densities were calculated with data collected from soil pit characterizations to a depth of 1 m)

City/county	Soil name/class.	Soil pit or location label	Type/land use	Carbon density (kg m^{-2})
Queens, NY	Gravesend	S97NY-081-008	Refuse	13.9
Richmond, NY	Greatkills	S95NY-085-006	Refuse	20.4
Kings, NY	Jamaica	S99NY-047-001	Dredge (old)	3.9
Kings, NY	Barren	S99NY-047-002	Dredge (old)	4.0
Kings, NY	Fortress	S99NY-047-003	Dredge (old)	4.5
Kings, NY	Big Apple	S95NY-047-001	Dredge (old)	2.9
Baltimore, MD[a]	n/a	S82MD-510-2	Dredge (recent)	24.7
Queens, NY	Verazano	S98NY-081-001	Clean fill (loamy)	28.5
Richmond, NY	Greenbelt	S95NY-085-033	Clean fill	3.6
Richmond, NY	Canarsie	S95NY-085-013	Clean fill	3.4
Richmond, NY	Central Park	S95NY-085-032	Clean fill	6.9
Washington, DC[b]	n/a	profile 2-4	Clean fill	1.4
Washington, DC[b]	n/a	profile 2-5	Clean fill	1.6
Moscow, Russia[c]	Urbanozem	Plot 23	Residential	12.9
Moscow, Russia[c]	Urbanozem	Plot 13	Residential	16.3
Chicago, IL[d]	n/a	Block 1	Residential	18.5
Chicago, IL[d]	n/a	Block 2	Residential	14.1

[a] Calculated from data reported in Evans et al. (2000) and data provided by D.S. Fanning.
[b] Calculated from data reported in Short et al. (1986).
[c] Calculated from data reported in Stroganova et al. (1998).
[d] Calculated to a depth of 60 cm. Data from Jo and McPherson (1995).

organic C density was higher among made soil types within a city than between cities for an individual soil type (Table 2). For example, within New York City the highest soil organic C density occurred on a golf course underlain by loamy fill (28.5 kg m^{-2}),while the lowest density occurred in an old dredge site (2.9 kg m^{-2}); an almost 10-fold difference. Comparisons of residential areas between Chicago (Jo and McPherson, 1995) and Moscow (Stroganova et al., 1998), however, showed relatively consistent soil C densities across the pedons included in our analysis (15.5 ± 1.2 kg m^{-2}). These residential soil organic C densities were close to four times the densities estimated for cropland in the Mid-Atlantic states, and were close to densities reported for Northeastern and Mid-Atlantic forests (Table 3).

Using these estimates of organic C densities of disturbed and made soil types found in urban areas, and assuming that on average roughly 60% of the land area of urban metropolitan areas are composed of these soils, we estimate that 26.3×10^{14} g and 10.7×10^{15} g of organic C exist in soils situated in urban ecosystems on a national (lower 48 states) and global basis (Table 4). These estimates are based on the average areal coverages of urban land-use calculated by Nowak et al. (1996) for the United States (lower 48 states). Our C density estimate of all soils found in urban ecosystems (8.2 kg m^{-2}) approximates C densities of soils developing in temperate thorn steppe life zones (Table 4). These calculations, however, are very preliminary and may vary considerably among cities within the USA and globally. Furthermore, our analysis shows that considerable variation can occur between soils in urban landscapes and therefore our C density estimate should not be considered representative of soil found in an urban ecosystem (Table 2).

4.3. Baltimore City extensive plots

Organic C amounts in exposed (i.e. not covered by impervious layers) surface soil (0–15 cm) varied widely across land-use types in Baltimore City (Fig. 4). Both C density (kg m^{-2} at 1 m depth) and concentration (g kg^{-1}) responded to land-use in the same way. This similarity was attributed to the absence of coarse fragment data, which may vary considerably across soils in these land-use categories (Fig. 4 and Table 5). In addition, soil organic C and organic matter data in this study were derived from loss on ignition, which may overestimate the amount of organic C in soils with high clay contents. Soil organic C was highest in low density residential and institutional land-use types; however, these areas exhibited the greatest variation in C density (Fig. 4). The higher variation may in part be due to the lower number of sampling points in these land uses. Nonetheless, low density residential and institutional land-use types had 44 and 38% higher organic C densities, respectively, than the commercial land-use type. Forest cover, medium density and high density residential, and transportation rights-of-way had intermediate organic C densities (Fig. 4).

To explore relationships between soil organic matter and the density of soil, we regressed bulk density with soil organic matter. Bulk density can serve as an indicator of soil trampling or other disturbance. The

Table 3
Comparison of soil C densities (kg m^{-2} at 1 m depth) for various "made" or disturbed soils with regional forest soil and cropland estimates of the Northeast and Mid-Atlantic states[a]

Land-use/region	Area ($\times 10^{10}$ m^2)	Carbon density (kg m^{-2})	Soil carbon ($\times 10^{14}$ g)
Northeast forest[b]	20.81	16.2	33.7
Northeast cropland[b]	–	6.0	–
Mid-Atlantic forest[b]	20.29	11.2	22.7
Mid-Atlantic cropland[b]		4.2	
Urban (residential)[c]	6.35	15.5 ($\pm$1.20)	9.84
Urban (undisturbed)[c]	7.34	9.4 ($\pm$1.40)	6.90
Urban (other)[c]	18.37	5.14 (?)	9.46
Old dredge		3.8 ($\pm$0.34)	
Refuse	–	17.2 ($\pm$3.34)	–
Clean fill	–	3.8 ($\pm$0.99)	–
USA urban (total)	30.06[b,d]	8.2 (?)	26.28
USA (total)	915.9[b,d]	6.8[e](?)	619.15

[a] Urban soil types (residential, old dredge, refuse, and clean fill) were compiled from soil pedon data presented in Table 2.
[b] C densities from Birdsey (1992) and aerial coverages calculated from Table 1, USDA Forest Service 1997 RPA report.
[c] Average areal coverage of urban land-use calculated from Nowak et al. (1996).
[d] Totals do not include Alaska or Hawaii.
[e] C density values from Birdsey (1992) and based on the proportion of land uses in the USA reported in World Resources Report (1996).

Table 4
Selected life zone areal coverages, soil carbon densities (kg m^{-2} at 1 m depth), and total soil carbon pools in comparison to urban land on a global basis

Group	Area ($\times 10^{12}$ m^2)	Carbon density (kg m^{-2})	Soil carbon ($\times 10^{15}$ g)
Life zone[a]			
Boreal forest-wet	6.9	19.3	133.2
Temperate forest-cool	3.4	12.7	43.2
Temperate forest-warm	8.6	7.1	61.1
Temperate thorn steppe	3.9	7.6	29.6
Temperate steppe-cool	9.0	13.3	119.7
Tropical forest-moist	5.3	11.4	60.4
Wetlands	2.8	72.3	202.4
Urban[b]	1.3	8.2[c] (?)	10.7 (?)
World total[d]			1500 ($\pm$20%)

[a] Data from Post et al. (1982).
[b] World urban land total from World Resources Institute (1996).
[c] Urban land soil C density estimate based on data presented in Table 3.
[d] World total estimate from Schlesinger and Andrews (2000).

regression revealed a significant negative relationship ($P<0.001$) with approximately 30% of the variation in organic matter concentration being explained by soil bulk density (Fig. 5).

5. Discussion

Our analysis of organic C pools in the New York City metropolitan area, City of Baltimore, and other cities suggest that urbanization has the potential to both directly and indirectly affect soil C pools. Our 10-year study of oak stands located along an urban–rural land-use gradient showed that abiotic and biotic environmental factors can substantially change as the adjacent land-use becomes more urbanized, and that these changes can affect soil chemistry, temperature regimes, soil community composition, and nitrogen and C fluxes (Groffman et al., 1995; Pouyat et al., 1995a, b). These indirect effects suggest that urbanization and the resultant environmental changes that occur can influence soil C pools even in forested ecosystems that are not directly or physically disturbed by urban development.

Our analysis of pedon data from several disturbed soil profiles suggest that physical disturbances and anthropogenic inputs of various materials (direct effects) can greatly alter the amount of C stored in these human "made" soils. Moreover, differences in surface soil C pools across several land-use types suggest that human activities, such as lawn maintenance practices, are strong controllers of variation in C dynamics in urban ecosystems. Below we discuss in more detail the implications of these preliminary findings on local, regional, and global scale soil organic C pools.

5.1. Effects of urban–rural environmental gradient

Environmental changes that have previously been measured along the urban–rural land-use gradient in the New York City metropolitan area appear to have affected forest soil organic C pools (Fig. 2). Many of these differences appear to be related to earthworm activity that occurs primarily in the urban stands (Steinberg et al., 1997). Where earthworms are present (mull soils), surface mineral soil (0–10 cm) organic C densities were higher than where earthworms were absent (mor soils). In contrast, earthworm activity has greatly reduced the O_2 layer, thereby decreasing overall forest floor organic C densities (Fig. 3). Ultimately, earthworms appear to affect only the distribution, and not necessarily the total amount, of C in these soils. While earthworms can accelerate organic matter decomposition, they also foster the production of soil aggregates, which can increase the physical protection, and storage of C (Martin, 1991; Scheu and Wolters, 1991). This reasoning is consistent with previous analysis of soil C pools along this transect, which suggests that pools of labile C are lower and pools of passive C are higher in urban relative to rural forest stands (Groffman et al., 1995).

In our analysis of soils not affected by earthworms, forest floor mass was higher in the urban core than in stands >40 km away (Fig. 1). This increase in forest

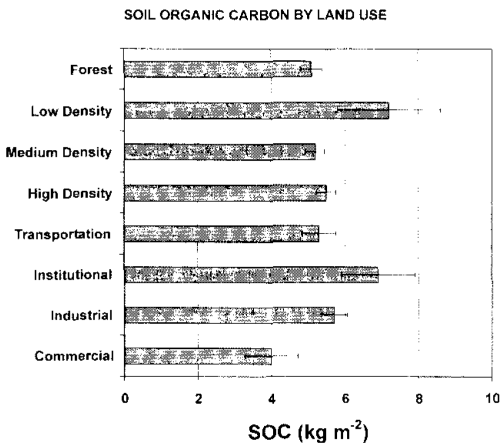

Fig. 4. Means (±S.E.) of organic soil C densities (kg m^{-2} at 15 cm depth) for land use types in the City of Baltimore. The number of plots used for calculating each mean is given in Table 5.

floor mass in non-earthworm affected stands in or near the urban core may in part be due to the input of lower quality leaf litter in these stands. Carreiro et al. (1999) found that red oak (*Quercus rubra* L.) litter collected from the rural stands decomposed more rapidly than litter from suburban and urban stands, in both field and laboratory incubations. An alternative explanation for the apparently high forest floor mass in the urban stands is differences in leaf litter inputs across the gradient. Pouyat (1992), however, found that leaf litter inputs in these stands were not significantly different along this gradient.

These results suggest that while urban conditions (presence of non-native earthworms, elevated soil temperature) tend to accelerate decay, urban litter quality may tend to decrease decay rates. The net result appears to be that when earthworms are present (mull soils), O_2 layers are denuded and mineral soil organic C pools are elevated due to the earthworm activity (Figs. 2 and 3). In contrast, when earthworms are absent (mor soils) from urban and suburban stands, forest floor mass is higher in urban than in the more rural stands, which may be due to the production of relatively low quality litter in stands in or near the urban core (Fig. 1).

The data presented in our analysis here and in previous studies suggest that in the absence of earthworms, forest stands exposed to urban environments have the potential to sequester and store more C than rural stands of the same canopy species composition. This pattern depends on consistent annual reductions in litter quality in urban areas. It is important to note that reductions in litter quality that increase passive C and decrease labile C pools can affect a variety of microbial processes important to ecosystem functioning. For example, we have observed lower microbial biomass (Groffman et al., 1995) and reduced rates of methane uptake (Goldman et al., 1995) in urban compared to suburban and rural stands. We are currently establishing studies along urban–rural transects in Baltimore City, MD and Budapest, Hungary to determine if the patterns we have observed in the New York City metropolitan area occur in other cities.

Table 5
Means (±S.E.) of soil bulk density and organic matter concentration (15 cm depth) by land-use type for 127 circular plots (11.35 m radius) in the city of Baltimore

Land-use	*n*	Bulk density (mg m^{-3})	Organic matter (g kg^{-1})
Forest	37	1.18 (0.04)	29.8 (1.8)
Low density residential	3	1.22 (0.03)	39.9 (8.6)
Medium density residential	29	1.18 (0.03)	29.7 (1.8)
High density residential	29	1.22 (0.03)	30.5 (1.7)
Transportation	5	1.17 (0.07)	30.8 (4.1)
Institutional	8	1.00 (0.11)	49.2 (8.5)
Industrial	3	1.41 (0.09)	27.4 (3.4)
Commercial	4	1.26 (0.16)	22.1 (4.5)

5.2. *Highly disturbed and made soils*

Disturbed and made soils exhibited a wide range in soil organic C amounts. Soil C densities ranged from a high of 28.5 kg m^{-2} in loamy fill material underlying a golf course in New York City to a low of 1.4 kg m^{-2} in clean fill materials deposited almost 100 years ago in the Mall in Washington, DC (Table 2). We had similar results for surface soil in the City of Baltimore, where mineral soil (0–15 cm) organic C concentrations and densities varied widely among land-use types (Fig. 4). On the other-hand, for any particular soil disturbance or fill type included in our analysis, the C densities were surprisingly similar. For example, residential sites in Chicago, IL and Moscow, Russia had very similar soil organic C densities (Table 2). Similarly, C densities for old dredge materials across four different sites varied by only 1 kg C m^{-2}. If these consistencies in soil C densities hold up for other urban ecosystems, it may be possible to use soil organic C data as a criterion in developing soil series concepts for highly disturbed and "made" soils (Pouyat and Effland, 1999; Hernandez, 1999).

Our comparisons of soil organic C density between "made" and forest soils suggest that residential areas have nearly the same C density as Northeastern forests and higher density than Mid-Atlantic forests (Table 3). This finding was unexpected, since many residential lawns are clipped during the growing season with organic material being removed in the process. On the other hand, lawns often receive high rates of nutrient inputs and water, which should increase above- and below-ground productivity. We suspect that the relatively high soil C amounts occurring in residential areas are primarily due to increases in belowground productivity. Lawns also have a much longer growing season than forests.

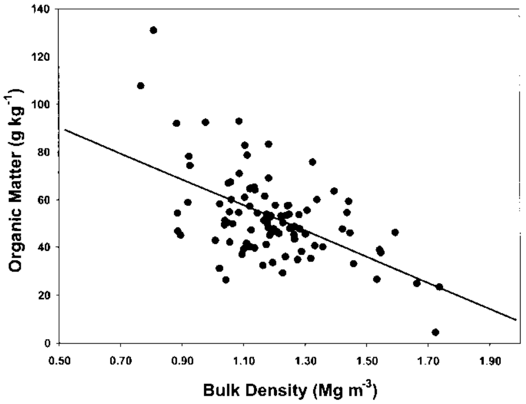

Fig. 5. Plot of soil organic C density (kg m^{-2} at 15 cm depth) as a function of soil bulk density for data collected in the City of Baltimore. Bulk density values are the mean of three cores. Soil organic C values are for one composite sample per plot (10–15 cores).

Obviously, lawn maintenance efforts vary widely within and across urban ecosystems and thus soil C pools should as well. Based on our analysis of the Baltimore City data, in those land-uses where lawn maintenance is expected to be relatively high, such as low density residential and institutional land-uses, we found the highest and most variable surface soil C densities (Fig. 4). It is interesting to note that while residential soil pedons in our analysis appear to be similar to Northeast Forests in C density on a regional scale, residential areas are more similar on a global scale to cool temperate steppe life zones (Table 4). This result is consistent with plant surveys of urban ecosystems where temperate steppe is a classification designated to describe vegetation structure (Dorney et al., 1984).

While residential soils are similar to forested and steppe ecosystems, the disturbed and "made" soils in our analysis had similar C densities to Northeastern and Mid-Atlantic croplands (Table 3). This result was not unexpected since cropland soils are highly disturbed with a large proportion of the biomass produced being harvested. However, like highly maintained lawns, these lands often receive large inputs of nutrients and water. The main difference between cropland and lawns would appear to be the physical disturbance from soil cultivation that reduce soil C levels (Paul and Clark, 1996). Indeed, for the land uses we sampled in the City of Baltimore, physical disturbances resulting in soil compaction (i.e. higher bulk densities), may be a factor affecting soil organic C pools (Fig. 5).

On a national scale, urban areas make up approximately 3.5% of the land base (Dwyer et al., 1998). Our estimate of 26.3×10^{14} g of soil organic C stored in urban ecosystems does not represent a significant proportion of the national (lower 48 states) total for soil C storage (Table 3). Likewise, on a global scale, urban areas make up approximately 1% of the land base and only 0.7% of the soil C pool (Table 4). For comparison, wetland soils which represent only slightly more than 2% of the world's land base, constitute the largest pool of soil organic C (Table 4). Although urban land-use conversions represent a relatively small proportion of the land base, we suspect the changes occurring in soil C storage will be more persistent than other land-use conversions.

Whether C storage is increasing or decreasing on a national basis as landscapes are urbanized will likely depend on the region of the country. For the cities included in our analysis, we have estimated an average C density of 8.2 kg m^{-2}, which for the northeastern United States would represent a decline in soil C storage relative to native soils prior to urbanization (Table 3). For the Mid-Atlantic States, however, this C storage estimate would represent an increase (Table 3). It is unclear what the net result would be in other regions of the country as our estimate of urban soil C density utilizes data mainly collected for cool to warm temperate areas. It would be interesting, for example, to compare residential soils in cities located in warmer and drier climates with those included in this study. We hypothesize that management inputs of water and fertilizer would lead to increases in soil C storage relative to native soils in warmer and drier climate zones. Obviously, more data is needed to test this hypothesis and to calculate more accurate estimates of soil C densities in urban ecosystems.

6. Conclusions

Based on our preliminary analysis of soil organic C pools of unmanaged forest stands, highly disturbed soils, and surface soils of various urban land-use types, urbanization can directly and indirectly affect soil C pools. Our analysis also suggests that soil C storage in urban ecosystems is highly variable with very high and low C densities (kg m^{-2} to a 1 m depth) present in the landscape at any one time. Although these urban effects may not yet have global significance relative to other life zones, it is important to note that changes associated with urbanization are more likely to persist than changes associated with many other land-use conversions. There is a strong need then to consider urban effects when calculating C budgets in localities and regions experiencing rapid urban expansion.

Indirect effects of urban environments on soil C are complex and variable. In oak stands along an urban–rural transect in the New York City metropolitan area, non-native earthworms and changes in litter quality were important. It is not clear how generalizable these factors are to other urban areas. These studies, therefore, need to be repeated in other cities located in similar and dissimilar life zones to the cities included in our analysis. Results from our gradient analysis, however, does 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 suggests that more data are needed on highly disturbed soils, such as land-fill, managed lawns, and covered soils to make regional and global estimates of soil C storage in urban ecosystems. Specific uncertainties include the quality of the C inputs (e.g. the quality of exotic plant species litter and stress effects on native species quality of litter), the fate of soil C in covered soils, measurements of soil organic C at depths greater than 1 m particularly in "made" soils, and spatially delineating disturbed and "made" soil types. Investigations also are needed to determine the sensitivity of decomposition to urban environmental changes, long-term effects of soil physical disturbances on C pools, and the effects of urban hydrological changes on wetland and riparian soils.

Acknowledgements

We thank J. Russell-Anelli and W. Zipperer and two anonymous reviewers for their comments on this manuscript. Funding support came from the USDA Forest Service, Northern Global Change Program and Research Work Unit (NE-4952), Syracuse, NY and the Baltimore Ecosystem Study's Long Term Ecological Research grant from the National Science Foundation (DEB 97-14835). The use of trade, firm, or corporation names in this publication is for the information and convenience of the reader. Such use does not constitute an official endorsement or approval by the US Department of Agriculture or the Forest Service of any product or service to the exclusion of others that may be suitable. This paper was presented at the USDA Forest Service Southern Global Change Program sponsored Advances in Terrestrial Ecosystem: Carbon Inventory, Measurements, and Monitoring Conference held 3–5 October 2000 in Raleigh, North Carolina.

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