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A comparison of soil organic carbon stocks between residential turf grass and native soil

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Abstract A central principle in urban ecological theory implies that in urbanized landscapes anthropogenic drivers will dominate natural drivers in the control of soil organic carbon storage (SOC). To assess the effect of urban land-use change on the storage of SOC, we compared SOC stocks of turf grass and native cover types of two metropolitan areas (Baltimore, MD, and Denver, CO) representing climatologically distinct regions in the United States. We hypothesized that introducing turf grass and management will lead to higher SOC densities in the arid Denver area and lower densities in the mesic Baltimore area relative to native cover types. Moreover, differences between turf grass soils will be less than differences between the native soils of each metropolitan region. Within Baltimore, turf grass had almost a 2-fold higher SOC density at 0- to 1-m and 0- to 20-cm depths than in rural forest soils, whereas there were no differences with soils of urban forest remnants. Moreover, urban forest remnants had more than 70% higher SOC densities than rural forest soils. Within Denver, turf grass (>25 years of age) had more than 2-fold higher SOC densities than in shortgrass steppe soils, while having similar densities to Baltimore turf grass soils. By contrast, the native soils of Baltimore were almost 2-fold higher than the native steppe grass soils of Denver using SOC densities of remnant forests as representative of native soils in the Baltimore region. These results supported our hypothesis that turf grass systems will be similar in SOC densities across regional variations in climate, parent material, and topography. These similarities are apparently due to greater management efforts in the Denver region to offset the constraint of climate, i.e., anthropogenic factors (management supplements) overwhelmed native environmental factors that control SOC storage.

Keywords Land-use change · Lawns · Soil carbon · Turf grass · Urban soils

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Introduction

An important focus of global climate change research is to address the question of how various land use and management changes will affect soil carbon (C) storage (Houghton et al. 1999; West and Six 2007). In particular, the importance of converting forests and grasslands to urban and suburban uses has recently become a focus of global change research (Pataki et al. 2006). Urban land use change entails a complex array of land and ecosystem alterations often missing in local, regional, and global estimates of soil organic carbon (SOC) stocks (Howard et al. 1995; Kaye et al. 2005; Pouyat et al. 2006).

A central principle in urban ecological theory implies that in urbanized landscapes anthropogenic drivers will dominate natural drivers in the control of SOC storage (Kay et al. 2006). The implication being that urban SOC stocks should converge on regional and global scales relative to the native systems being replaced, i.e., the Urban Ecosystem Convergence hypothesis (Pouyat et al. 2002; 2003). The convergence in SOC stocks occurs as long as the anthropogenic drivers (management and use), and their effects on SOC dynamics dominate native controlling factors, e.g., topography and parent material. The change occurring in SOC stocks as forests and grasslands are converted to urban and suburban land uses integrate a number of ecosystem attributes, such as net primary productivity, or C input, and organic matter decay, or C output, and can thus serve as an integrated measure of ecosystem response to urban land-use change.

An important characteristic of urban land-use conversion with respect to SOC storage is the introduction of turf grass cover and management (Kaye et al. 2005; Milesi et al. 2005; Golubiewski 2006). Approximately 41% of the total urban area in the conterminous United States is in residential use, most of which is dominated by turf grass (Nowak et al. 1996, 2001). With approximately 3.5% to 4.9% of the nation's land base in urban use (Nowak et al. 2001; National Association of Realtors 2001), the total estimated areal amount of turf grass for the conterminous United States is 163,800±35,850 km^2, which exceeds by three times the area of irrigated corn (Milesi et al. 2005). To manage turf grass areas, homeowners and institutional land managers apply about 16 million kilograms of pesticides each year (Aspelin 1997) as well as fertilizers at rates similar to or exceeding those of cropland systems, e.g., up to 200 kg ha^{-1} $year^{-1}$ (Law et al. 2004). In addition, turf grasses are typically irrigated and clipped regularly during the growing season.

Although lawn care practices add chemicals at rates comparable to cropland systems, they are potentially less disruptive to C dynamics (Pouyat et al. 2003). Cropland systems have a greater magnitude and frequency of soil disturbances and generally remove a greater proportion of the standing crop (Asner et al. 1997). As a result, cropland systems can lose substantial amounts of SOC (Post and Mann 1990; Matson et al. 1997), whereas turf grass ecosystems can accumulate SOC at rates similar to or greater than those for grasslands and forests because of the absence of annual soil disturbances and the addition of supplements such as water and fertilizer (Qian and Follett 2002; Pouyat et al. 2003). In comparing surface soils of 15 golf courses, Qian and Follett (2002) found that total C accumulation rates ranged from 0.9 to 1.0 t C ha^{-1} $year^{-1}$ and reached a maximum between 30 and 50 years. This rate is higher than unmanaged (0.33 t C ha^{-1} $year^{-1}$) (Post and Kwon 2000) and semiarid (0.03 t C ha^{-1} $year^{-1}$) (Burke et al. 1995) grasslands, but similar to rates obtained for perennial grasslands following cultivation (1.1 t C ha^{-1} $year^{-1}$) (Gebhart et al. 1994). Using long-term simulations of the CENTURY model for lawn ecosystems, Qian et al. (2003) showed that N fertilization coupled with the replacement of grass clippings increased SOC accumulations by up to 59% in comparison to sites that were not fertilized and where clippings were removed. Likewise, Higby and Bell (1999) found that soil organic matter was higher in fertilized golf course fairways than in adjacent unfertilized areas. Golubiewski (2006) measured SOC stocks in 13 residential yards of different ages built on the semiarid shortgrass of Colorado and found that SOC recovered from the initial disturbance caused by development after 25 years. Moreover, the effects of lawn management on SOC accumulation exceeded the effects of other soil-forming factors (*sensu* Effland and Pouyat 1997; Pickett and Cadenasso, this vol.), such as elevation and soil texture.

Based on these findings, the amount of effort to manage turf grass systems, e.g., intensity of irrigation or nutrient applications is an important determinant of SOC accumulation post-urban development. Furthermore, the intensity of effort is dependent on the natural constraints of the system. For example, irrigation rates for turf grass growing in dry land areas will be higher than for turf grass in more temperate climates. Indeed, in the absence of irrigation and fertilization, a majority of species of turf grass would not be able to grow in most of the United States (Milesi et al. 2005). Therefore, as the constraints to the growth of turf grass increase, the management effort will also have to increase so that turf grass species can grow and compete favorably with other plants. The maintenance of highly productive turf grass cover should result in a higher accumulation of SOC than in the previous dry land systems (e.g., Golubiewski 2006; Jenerette et al. 2006). In temperate regions where turf grasses may grow without supplements for most of the growing season, SOC in residential areas should be equivalent to or lower than the native forest soil, which has potentially higher biomass inputs (unmanaged forest vs. a mowed lawn) and more recalcitrant organic material (e.g., wood) entering the soil surface (Jobbágy and Jackson 2000).

In the present study, we combined measurements of SOC stocks in two metropolitan areas (Baltimore, MD, and Denver, CO) representing climatologically distinct regions of the United States (temperate hardwood forest and semiarid shortgrass steppe) to compare residential turf grass with native SOC stocks of each area. We hypothesized that the difference between soils of residential turf grass and native cover types will depend on the characteristics of the native or rural ecosystem replaced. For the more arid region (shortgrass steppe) associated with the Denver metropolitan area, the effect of converting native soils to residential use (i.e., introduction of turf grass and management) will lead to higher SOC densities, whereas in more humid environments (hardwood deciduous forest) associated with the Baltimore area there will be no effect or a slight reduction in SOC from converting to turf grass. Furthermore, if impervious surfaces are accounted for, i.e., the net change due to converting to urban land use, Denver will be slightly higher and Baltimore lower in SOC density than the previous native landscape. Moreover, because of regional differences in climate, resource constraints, and plant cover, differences in SOC densities among native soils (arid grassland steppe vs. temperate deciduous forest) will be greater than the differences among residential turf grass soils. Finally, in addition to differences in SOC densities, we expected the percentage of SOC in the top 20 cm (relative to 1 m) be greater in tree-dominated cover types than in grassland cover types, as reported for soils on a global scale by Jobbágy and Jackson (2000), and that turf grass systems will in the long term approximate the SOC distribution of native grassland systems.

Site descriptions

Baltimore, MD

Baltimore City is a historically industrial city located on the Chesapeake Bay in the Mid-Atlantic region of the USA (39°17′11″ N, 76°36′54″ W). Between 1950 and 2000, Baltimore's population fell significantly from 949,708 to 638,251 (http://quickfacts.census.gov/qfd/states/24000.html). The city's depopulation has resulted in a large number of residential housing vacancies and vacant lots. Most of the housing stock in Baltimore was built before 1950 in contrast to the surrounding Baltimore County area, where housing growth increased substantially after 1950. One of the study sites, Cub Hill, is a suburban neighborhood located in Baltimore County 14 km northeast of Baltimore city center and is predominately made up of medium density residential lots built after 1970.

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

Baltimore lies between two physiographic provinces: the Piedmont Plateau and the Atlantic Coastal Plain. The north–northeast trending Fall Line separates the two provinces, dividing the city approximately in half. Most of the city is characterized by nearly level to gently rolling uplands, dissected by narrow stream valleys. Soils in the Piedmont Plateau of Baltimore are very deep, nearly level to moderately sloping, well-drained upland soils underlain by semi-basic or mixed basic and acidic rocks. Soils in the Coastal Plain of the city are very deep, somewhat excessively drained and well-drained upland soils underlain by either sandy or gravelly sediments or unstable clayey sediment (NRCS 1998). Highly disturbed soils make up more than 60% of the land area of the city (Pouyat et al. 2002). Cub Hill is underlain primarily by the Joppa soil series and associated soil types. These soils consist of deep, well-drained to somewhat excessively drained soils on the uplands of the Coastal Plain. They formed in mostly old sandy and highly gravelly deposits (NRCS 1998).

The Baltimore metropolitan area was previously dominated by hardwood deciduous forests with smaller areas of riparian and wetland soils (Schneider 1996). After European colonization and before the development of the city, the forested areas were transformed to agricultural uses. Forest cover, mostly outside the city, is dominated by chestnut oak (*Quercus prinus* L.), underlain by coarser soils weathered from schist; yellow-poplar (*Liriodendron tulipifera* L.) occurring on thicker saprolitic soils weathered from gneiss and granite; and boxelder (*Acer negundo* L.) green ash (*Fraxinus pennsylvanica* Marsh.), sycamore (*Platanus occidentalis* L.), and silver maple (*Acer saccharinum* L.) in riparian areas (Brush et al. 1980). The remnant forest at Cub Hill is dominated by yellow-poplar (*L. tulipifera*), white oak (*Quercus alba* L.), and various hickory species (*Carya* spp.). Overall, Baltimore City has a canopy cover of approximately 25% with the majority of tree stems occurring in remnant forest patches, vacant land, and residential areas dominated by ash species (*Fraxinus* spp.), American elm (*Ulmus americana* L.), American beech (*Fagus grandifolia* Ehrh.), black cherry (*Prunis serotina* Ehrh.), black locust (*Robinia pseudoacacia* L.) and the tree-of-heaven (*Ailanthus altissima* Mill.) (Nowak et al. 2004).

Denver/Boulder, CO

The Denver metropolitan area has experienced rapid and expansive development patterns throughout the past decade (Golubiewski 2006). The region is comprised of a matrix of urban, agricultural, and grass land use and cover. The Denver metropolitan area is situated on the Colorado Front Range at the base of the foothills on the western edge of Colorado's plains (~40° N, ~105° W). Since the mid-1800s, dry land and irrigated farming have transformed much of the region's natural grasslands. Urbanization followed in the early twentieth century and increased markedly midcentury through the 1990s, so that by 2000 Denver's metropolitan area reached a population of 2.2 million people (http://www.census.gov/population/cen2000/phc-t29/tab04a.pdf). The eastern plains of Colorado, including the prairie portion of the Front Range, are comprised of semi-arid shortgrass steppe (Sims and Singh 1978) with mean annual precipitation of 30 to 40 cm and mean annual temperature of 8°C to 10°C. Most (70–80%) precipitation falls during the growing season (April–September) with heavy rainfalls occurring during monsoon season in July and August (Doesken et al. 2003).

Sampling locations were scattered throughout the Denver–Boulder metropolitan area in suburban/urban cover types. These locations corresponded to homeowners who volunteered to participate in the "Front Range Neighborhood Program" and allowed data collection on their properties (Golubiewski 2006).

Methods

In Baltimore, suburban sites were located within the city and in a suburban neighborhood (Cub Hill) just outside the city boundary. In Denver, suburban areas were located outside the city throughout the metropolitan area. In each case, comparisons of SOC densities at different depths were made of residential turf grass and native cover types that were likely the precursors to the residential areas sampled. We acquired data to calculate SOC densities (kg m^{-2}) of turf grass and native soil to depths of 0- to 20-cm and 0- to 1-m in both metropolitan areas using three sources: (1) a meta analysis of data from two previous studies (Golubiewski 2006; Pouyat et al. 2006); (2) pedon data in the National Resource Conservation Service (NRCS) National Soil Characterization database (http://ssldata.nrcs.usda.gov/default.htm); and (3) new field measurements. By summarizing the data at two depths (0- to 20-cm and 0- to 1-m), we were able to calculate the proportion of SOC occurring in the top 20 cm of each 1-m pedon, which should reflect the dominant plant cover and use of individual soils (Jobbágy and Jackson 2000). Finally, to estimate the net effect of urban land-use change on SOC densities in both metropolitan areas, we calculated SOC densities on an area-weighted basis (including impervious areas) for pre-urban, agricultural, and post-urban landscapes.

Baltimore, MD

To measure SOC in Baltimore, we used 1-m undisturbed cores. The core method was used in lieu of excavating a soil pit because it is less intrusive and allows for more replications at each location (Pouyat et al. 2006). Within Baltimore City, we located plots within medium density residential areas and in relatively undisturbed remnant forest patches (heretofore referred to as urban remnant forests) occurring primarily on city-owned parkland on the Piedmont Plateau. In the Cub Hill neighborhood, we sampled medium density residential lots and a 2.0-ha remnant forest patch (heretofore referred to as suburban remnant forest). The locations within Baltimore City were randomly selected from 202 0.04-ha circular plots that were sampled for vegetation and surface soils in previous studies (Nowak et al. 2004; Pouyat et al. 2007). A total of 19 residential lots and five urban remnant forests were sampled to a 1-m depth. The majority of residences within the city were built before 1950 and only one residence was built after this date (1957). All of the urban forest remnants were at least 80 years of age.

At Cub Hill, we randomly selected seven residences out of a possible 33 from two subdivisions built in different years (1970 and 1980). Portions of each subdivision fell within a 1-km^2 area around an eddy flux tower operated by the US Forest Service as part of the Baltimore Ecosystem Study. At six of the residences a 2×2 m plot was established in a representative location in the front and back of the house while only the front of a seventh residence was sampled (total of 13 plots). In the adjacent suburban forest remnant, we collected cores from 2×2 m plots established along two transects running perpendicular to the length of a 10° to 15° slope. Three plots were established along each transect (top, middle, and bottom slope positions) for a total of six plots.

For all plots we extracted three 3.3-cm diameter cores (1-m depth) in a triangle at least 1.5 m apart around an approximated center point. Each core was brought to the lab for characterization and subsampled by horizon. For each horizon, bulk density was measured using the clod method (Blake and Hartge 1986). The proportion of coarse fragments was determined by passing a known weight of each subsample through a 2-mm sieve. Subsamples of soil were analyzed for total organic C using a Perkin Elmer 2400 CHNS Analyzer and a Shimadzu TOC Analyzer for the Baltimore City and Cub Hill samples, respectively. The samples were first ground and passed through a 2-mm sieve and subsequently pulverized by continuously rotating soil subsamples for at least 24 h in glass bottles containing steel rods.

Denver and Boulder

Soil was sampled at 13 sites (out of 53) that participated in the Front Range Neighborhood Program. More detailed information on original study design can be found in Golubiewski (2006). Soil sampling consisted of randomly collecting three soil cores from each yard. Soil cores were removed in 10-cm increments to a depth of 30 cm with an AMS hammer corer (ring diameter of 4.7 cm), which differed from the samples taken by horizon in the Baltimore 1-m cores. For a few cores, it was not possible to extract the 20- to 30-cm increment because of high soil densities. The complete soil dataset with all replicates consisted of 194 samples. Details of soil sample preparation are in Golubiewski (2006). Total C was measured by dry combustion on a Fisons CHN Analyzer (Nelson and Sommers 1996). Inorganic C was also determined by the acetic acid dissolution method, including a standard addition of 0.1 M $CaCl_2$ to address potential non-specific interactions of H^+ with clay minerals (Loeppert and Suarez 1996). Organic C was calculated by difference of total C and inorganic C. Bulk density was calculated using the core method (Blake and Hartge 1986; Culley 1993). For each sample, total bulk density was determined as the total grams of soil (constant oven-dry mass at 105°C) per cubic centimeter (calculated from the hammer corer ring). Fine bulk density was calculated as the oven-dry mass per volume of the fine fraction (i.e., the volume of the coarse fraction was subtracted from total volume of the soil core). The fine bulk density figures were used to scale soil C contents to a weight per area basis.

Determination of native soil SOC

Data for native soils were obtained from the NRCS Soil Characterization database (http://ssldata.nrcs.usda.gov/default.htm). In an effort to keep factors affecting soil formation as constant as possible, all pedon data occurring on the Piedmont of Maryland, under forest cover, and at similar slope and elevation positions were used to calculate the native SOC densities of the Baltimore metropolitan region. Similarly for Denver, pedon data from several counties of the Colorado Front Range (Kit Carson, Lincoln, Logan, and others) were located between the elevations of 1,150 and 1,940 m and similar slope positions and were used to calculate SOC densities of the semiarid shortgrass steppe cover type. We included only pedon data that characterized mineral horizons with measurements of coarse fragment percentage, oven dried bulk density, and organic C concentration. For the native soils of Maryland, we included pedons with measurements of total C because these soils did not possess high amounts of carbonate. In cases where data were not available to 1 m, we extrapolated to this depth if a C horizon was reached in the pedon characterization. As a supplement and reference to the pedon data of native soils, we compiled from the literature SOC densities representing northeastern Colorado's shortgrass steppe and the Piedmont of Maryland (Table 1). Furthermore, we considered direct comparisons between the remnant forest soils with residential soils in Baltimore and Cub Hill as a turf grass and native soil contrast.

Table 1 Soil organic carbon densities (kg m^{-2}) from the literature by vegetation and land unit cover occurring in the Baltimore, MD, and Denver, CO, metropolitan areas

Source	Cover/land unit	Depth (cm)	C density (kg m^{-2})	*n*	Elevation (m)	Location
Kaye et al. (2005)	Corn	0–15	1.66	3	1,493–1,620	Colorado
	Corn	15–30	1.86	3	1,493–1,620	Colorado
	Wheat	0–15	1.85	3	1,493–1,620	Colorado
	Wheat	15–30	1.66	3	1,493–1,620	Colorado
	Shortgrass prairie	0–15	2.85	3	1,493–1,620	Colorado
	Shortgrass prairie	15–30	1.93	3	1,493–1,620	Colorado
Yonker et al. (1988)	Lowlands	0–100	7.30	18	1,220–1,830	Colorado
	Slopes	0–100	5.20	12	1,220–1,830	Colorado
	Uplands	0–100	6.60	13	1,220–1,830	Colorado
Schimel et al. (1985)	Footslope	0–85+	7.98	3	1,641	Colorado
	Backslope	0–68+	3.81	3	1,647	Colorado
	Summit	0–53+	2.58	3	1,653	Colorado
Burke et al. (1990)	Swale	0–20	3.39	n/a	~1,000	Colorado
	Midslope	0–20	1.63	n/a	~1,000	Colorado
	Ridgetop	0–20	1.70	n/a	~1,000	Colorado
Gill et al. (1999)	n/a	Surface per 1 cm	0.14	8	1,220–1,830	Colorado
	n/a	75–100 per 1 cm	0.04	8	1,220–1,830	Colorado
Birdsey (1992)	Cropland	0–100	2.84	n/a	n/a	Rocky Mountain
	Grassland	0–100	3.81	n/a	n/a	Rocky Mountain
	Cropland	0–100	4.17	n/a	n/a	Mid-Atlantic
	Timberland	0–100	11.56	n/a	n/a	Mid-Atlantic
Bolstad and Vose (2005)	Deciduous forest	0–30	1.82	4	670–865	Southern Appalachian Mountains

n/a Not applicable

Data reporting and statistical analysis

For all data, the density of C in a horizon of unit area (1 m^2) was calculated as

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

where c is carbon density, $\delta_{2\ mm}$ is the fraction of material larger than 2 mm diameter, $\boldsymbol{B}_D$ is bulk density, c_f is the fraction by mass of organic C, and $\boldsymbol{V}$ is the volume of individual horizons (Post et al. 1982). Data for the soil horizons were added to report soil C density on a kilogram per square meter basis to depths of 20 cm and 1 m.

In Baltimore, for those cases where we were unable to extract a core to 1-m depth, we extrapolated from the lowest depth of the C horizon to reach 1 m. In those cases in which a C horizon was not reached, we excluded the core from the analysis. With these extrapolations we found no relationship between overall SOC density and the difference in length between the actual depth of the core and 1 m. This suggests the length of extrapolation is not related to the overall estimated SOC density value to the depth of 1 m (Pouyat et al. 2006). In Denver, SOC densities were extrapolated from the original depth of sampling (0 to 30 cm) to a 1-m depth by using proportional distributions of SOC measured in grasslands (Jobbágy and Jackson 2000). For grasslands, Jobbágy and Jackson reported the proportion of C in the 0- to 20-cm depth as 0.42 and in the 20- to 40-cm depth as 0.23 relative to 1 m. Assuming the C in the 20- to 40-cm depth is distributed evenly, we added half of this proportion (0.115) to that found in the 0- to 20-cm depth to derive an estimate of the proportion of C found in the 0- to 30-cm depth profile (i.e., 0.535). This proportion estimate was then used to extrapolate the field measurements to 1 m. In other words, the field measurement made up 53.5% of the C in the 1-m profile, and therefore the remaining 46.5% of C was added to the field measurement to estimate the SOC density to a 1-m depth.

With respect to statistical comparisons, we acknowledge that data were derived from sampling locations chosen for different purposes and based on availability (e.g., it was not always possible to sample on private property). Thus, the data do not represent a random sampling from a homogeneous population. Therefore, estimates based on the sample locations may not accurately reflect characteristics of all residential or native soils within a metropolitan area. However, even with an arbitrary sampling design, the sampling locations were selected without preconceived bias (McCune and Grace 2002) justifying the use of statistical comparisons to determine the potential for differences in the sample population.

To make comparisons within each metropolitan area, we used a nonparametric Kruskal–Wallis test (Proc NPAR1WAY, SAS Version 9.1, 2003). Statistical comparisons were made between residential and urban forest remnants in Baltimore City, between two subdivisions and adjacent suburban remnant forest patch at Cub Hill, and among residential lots of different ages in Denver. Comparisons among two or more medians between metropolitan areas constituted two different populations, which also required a nonparametric Kruskal–Wallis test. These statistical comparisons included the residential sites in Baltimore and Denver and the native soils representing shortgrass steppe in the Front Range and hardwood forests of the Mid-Atlantic states.

Finally, we made a simple calculation to illustrate the potential net change in SOC densities on an area-weighted basis (includes impervious surfaces) between pre- and post-urban landscapes of Baltimore and Denver. To do this, we obtained the percentage of impervious and pervious cover for Denver (Nowak et al. 1996) and Baltimore (Pouyat et al. 2006) and multiplied each by a SOC density assigned to each. In both cities, we assumed a SOC density of 3.3±0.9 kg m^{-2} for soils beneath impervious surfaces, or the density of clean fill calculated using data acquired from several urban areas (Pouyat et al. 2006). We assumed the pervious areas (1-proportion of impervious area) of each city were primarily covered with turf grass of >30 years of age, and thus assumed a SOC density of 11.0±0.9 and 12.7±1.3 kg m^{-2} for Baltimore and Denver, respectively (refer to results for the calculation of these values). With these assumptions, we compared the area-weighted SOC density of each city to densities in the soils of native forest, grass, or agricultural land that was likely present before the development of each city.

Results and discussion

Baltimore

Within Baltimore there were statistically significant differences in SOC density between cover types at 0- to 1-m depth, 0- to 20-cm depth, and the percentage of SOC in the top 0 to 20 cm (Fig. 1). Residential turf grass had a greater than 1.5-fold higher SOC density at 0- to1-m and 0- to 20-cm depths than in rural forest soils ($P<0.001$ and $P=0.053$), whereas there were no differences with soils of urban forest remnants (Fig. 1). However, differences were found for the percentage of SOC in the top 0 to 20 cm between residential turf grass and urban forest remnants ($P=0.016$). Urban forest remnants had more than 80% higher SOC densities at 0- to 1-m depth than rural forest soils ($P=0.028$).

Soil organic carbon densities of urban forest remnants were consistent with those reported in the literature for Mid-Atlantic hardwood forests (11.56 kg m^{-2}, Table 1). By contrast, rural forest SOC densities (0- to 1-m depth) reported in the NRCS database (6.7±0.8 kg m^{-2}) tended to be lower than those reported in the literature. Soil organic carbon densities of residential turf grass in Baltimore are much higher than those reported for urban fill (3.3± 0.93 kg m^{-2}), but slightly lower than other residential turf grass data (15.5±1.2 kg m^{-2}) reported by Pouyat et al. (2003) from the literature.

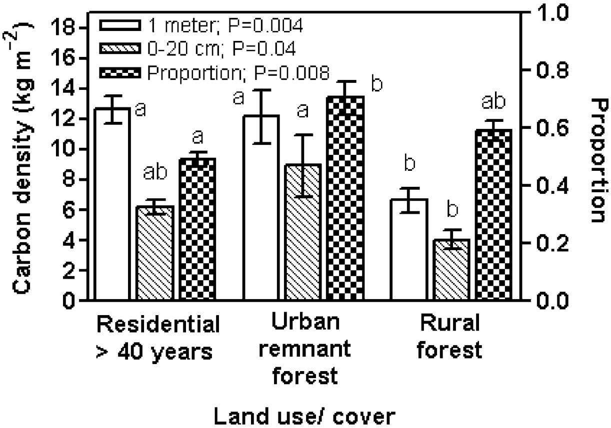

Fig. 1 Mean (±SE) SOC densities (1-m and 0- to 20-cm depths) and the percentage of SOC in the top 0 to 20 cm of residential ($n=19$), urban forest remnant ($n=5$), and rural forest ($n=7$) soils in the Baltimore, MD, metropolitan area. Soil organic carbon densities (1-m depth) of residential and urban forest remnants measured from undisturbed cores, and rural forest densities are from the NRCS National Soil Characterization database. Comparisons of SOC medians were done with a Kruskal–Wallis test (P values in graph). Medians (means shown) with *different letters* were significantly different

At the Cub Hill site, there were no significant differences between the suburban forest remnant and residential subdivisions in SOC densities at a 0- to 1-m depth and the percentage of SOC in the top 0 to 20 cm. However, differences of SOC at 0 to 20 cm were found between residential turf grass and the suburban remnant forest (Fig. 2). Specifically, at the 0- to 20-cm depth the suburban remnant forest had almost 2-fold higher SOC densities than in the 1970 and 1980 subdivision turf grass soils (P=0.004 and 0.007, respectively). Comparing all turf grass soils at 0- to 1-m depth, the older Baltimore residences had 88% and 66% higher SOC densities than in the 1970 and 1980 subdivisions at Cub Hill, respectively (Figs. 1 and 2). These differences were almost 2-fold higher at the 0- to 20-cm depth. The SOC densities of Baltimore City and Cub Hill remnant forests were similar (12.1±1.8 and 10.5±1.5 kg m^{-2}, respectively) and did not differ statistically (Figs. 1 and 2). Means for both forested areas were almost 2-fold higher than for forest soils occurring in the Piedmont of Maryland as reported in the NRCS database (Fig. 1), but slightly lower than for timberland soils (11.56 kg m^{-2}) reported in the literature (Table 1).

These results suggest the older turf grass soils in Baltimore City (>40 years) accumulated more SOC than the younger turf grass at Cub Hill (<35 years), which is consistent with Qian and Follett (2002) who found that maximum SOC accumulations in turf grass systems occurred between 30 and 50 years. Moreover, this length of time to reach an asymptote in SOC accumulation corresponds with that reported for grassland soils (Conant et al. 2001). If turf grass soils reach a maximum in SOC accumulation at approximately 40 years, the Cub Hill turf grass soils need an additional 10 to 20 years of organic C accumulation to reach the level of the Baltimore residential soils (Figs. 1 and 2). This would require an accumulation of more than 4 kg m^{-2} of C in one to two decades (0.4 to 0.2 kg m^{-2} $year^{-1}$, respectively), which represents a high rate of accumulation. Otherwise, relatively low SOC densities for Cub Hill residential soils may reflect the legacy of the native soil in the Cub Hill area (Joppa soil series), which developed in sand deposits in the upland Coastal Plain (NRCS 1998). The suburban forest remnant also occurs on the Joppa soil series and had a lower mean SOC density than the urban forest remnants in Baltimore City, although this difference was not statistically significant (Figs. 1 and 2). Finally, our selection of residential sites did not account for differences in lawn management, which can be quite large (e.g., Law et al. 2004) and thus affect SOC accumulations among residences (e.g., Qian et al. 2003; Milesi et al. 2005).

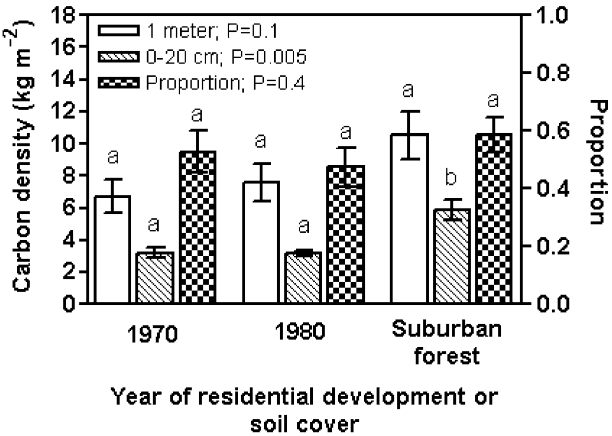

Fig. 2 Mean (±SE) SOC densities (1-m and 0- to 20-cm depths) and the percentage of SOC in the top 0 to 20 cm of residential turf grass (n=7 and n=8 for 1970 and 1980 subdivisions, respectively) and suburban forest remnant (n=6) soils in Cub Hill, Baltimore County, MD. Comparisons of SOC medians were done with a Kruskal–Wallis test (P values in *graph*). Medians (means shown) with *different letters* were significantly different

The reason for differences in SOC between the urban and suburban forest remnants and that of rural forest soils in the NRCS characterization database is not clear. The urban and suburban forest remnant SOC densities from this study approximated estimates for timberland in Maryland and the Mid-Atlantic states (Birdsey 1992; Table 1), which were calculated using a model based on Post et al. (1982). A potential, but unlikely, cause of the differences in SOC measurements between our data and data from the NRCS database may be differences in methodology. The coring technique used in this study may have overestimated bulk density measurements due to compaction that occurs when extracting the core, whereas pedon characterizations rely on clods extracted from the pit face (NRCS 2004). To make up the average difference between the forest remnant and rural forest SOC densities (4.5 kg m^{-2} or the difference between 11.2 and 6.7 kg m^{-2}) would require a 67% overestimation of the core bulk density measurements.

Another possible explanation for differences in forest SOC data may be differences in site history and stand age, information not available in the NRCS characterization database. For example, the NRCS database for Maryland is biased towards agricultural areas where second growth forests are likely to be growing on soils that were heavily cultivated or eroded under previous use. This is highly probable in the Piedmont of Maryland (Schneider 1996). Moreover, based on personal observation, forest remnants in urban and suburban areas tend to be older than rural forests because agricultural abandonment occurred earlier in more densely populated areas of the state. The urban and suburban forest remnants we sampled in this study were at least 80 years of age, suggesting that these forests may continue to accumulate SOC for 20 or more years (Post 2003). The age of forest associated with the NRCS data is unknown, but based on the previous arguments these forests are more likely younger than forest remnants in urban areas.

Based on the NRCS data comparison, our hypothesis that residential turf grass soils would have SOC densities similar to or lower than those for native forest soil types in the Baltimore metropolitan area is not supported. The accumulation of SOC in older residential areas of Baltimore clearly exceeds or is equivalent to forest soils in the region, although the rural forests included in this analysis may not have reached their maximum SOC accumulation. If we test our hypothesis based on comparisons between the urban and suburban remnants and the residential soils of the Baltimore metropolitan area, our hypothesis is supported (Figs. 1 and 2). In addition to the overall amount of SOC, the depth of accumulation appears to differ between the two cover types. For example, the urban forest remnants accumulated approximately 70% of SOC in the top 0 to 20 cm (relative to the first meter) compared to only 50% in the residential turf grass areas (Fig. 1). These results are consistent with results from Jobbágy and Jackson (2000), who analyzed data from more than 2,700 soil profiles worldwide and found that forest soils accumulated a greater proportion of SOC in the top 0 to 20 cm compared to grassland and shrubland vegetation types. Indeed, temperate forests have been shown to have higher aboveground allocation than temperate grasslands (Jackson et al. 1996; Cairns et al. 1997), which result in a higher accumulation of organic matter at the surface of the soil (Jobbágy and Jackson 2000).

Denver

Soil organic carbon densities of residential turf grass soil of >25 years were at least 30% higher than turf grass soils of residences built in the 1990s and native shortgrass steppe soils reported in the NRCS soil characterization database (Fig. 3). For each pairwise comparison between the 1990 and the 1970 and 1960 residences, the differences were statistically significant (P=0.025 and 0.050, respectively). Relative to the native shortgrass steppe soils, only the 1990 residences were not statistically higher in SOC density (Fig. 3).

The comparison of >25 year old residential soils to native shortgrass steppe soils suggests that urban land-use change can increase SOC densities by almost 2-fold in the Front Range of Colorado. This observation is consistent with Kaye et al. (2005), who found significantly higher belowground allocation of carbon in managed turf grass areas compared to shortgrass steppe and agricultural systems in the Fort Collins, CO, area (Table 1). In contrast to the Maryland NRCS native forest soil data, the shortgrass steppe data (7.3±0.53 kg m^{-2}) are consistent with data in the literature (Table 1). Yonker et al. (1988) reported a range of SOC densities of lowlands, mid slopes, and uplands of 7.3, 5.2, and 6.6 kg m^{-2}, respectively, in the Front Range of Colorado. Similarly, Schimel et al. (1985) measured a range of SOC densities between 2.6 and 8.0 kg m^{-2} and an area weighted mean of 6.4 kg m^{-2} in the Colorado Front Range. In both studies, the most important factor in SOC density was the geophysical unit and topographical position. These results support our hypothesis that turf grass soils would accumulate a higher amount of SOC than the semiarid shortgrass steppe soils of the Colorado Front Range (Fig. 3).

Comparison among the different aged subdivisions suggests that soils disturbed from residential development continued to accumulate SOC up to 40 years (Figs. 1 and 3). These observations are consistent with Qian and Follett (2002) who found maximum SOC accumulation (0 to 10 cm) occurring at 30 years for fairways and up to 50 years for putting greens. By contrast, Golubiewski (2006) reported SOC densities for 0- to 30-cm depth for the same residential plots in this study that showed a maximum accumulation among different aged residences of approximately 25 years. When we combined data from 1950 and 1960 aged subdivisions, a more gradual increase approximating 7 kg m^{-2} C over a 40 year period was revealed in this study (Fig. 3). If we assume a constant accumulation of SOC over this period that would mean an average increase of 0.18 t C ha^{-1} $year^{-1}$ for turf grass compared to 0.03 t C ha^{-1} $year^{-1}$ for shortgrass steppe soils reported for eastern Colorado (Burke et al. 1995). If these results are typical of residential soils throughout the

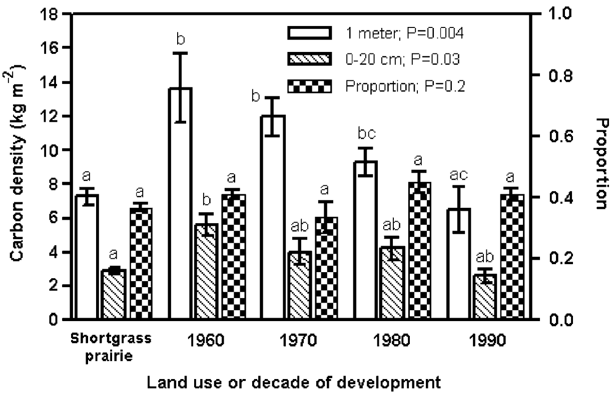

Fig. 3 Mean (±SE) SOC densities (1-m and 0- to 20-cm depths) and the percentage of SOC in the top 0 to 20 cm of residential (n=3, 2, 5, and 3 for 1990, 1980, 1970, and 1960 subdivisions, respectively) and shortgrass steppe (n=19) soils in the Denver metropolitan area. Residential SOC densities to 1-m depth extrapolated from 0- to 30-cm undisturbed cores, and shortgrass steppe densities from the NRCS National Soil Characterization database. Comparisons of SOC medians were done with a Kruskal–Wallis test (P values in *graph*). Medians (means shown) with *different letters* were significantly different

Colorado Front Range, then it would appear that turf grass cover and management resulted in a higher rate of SOC accumulation, but with a period of maximum accumulation (40 years) that is similar to native grasslands in the region (Burke et al. 1990).

Baltimore–Denver comparison

Residential turf grass soils of the Baltimore (including Cub Hill) and Denver metropolitan areas had similar SOC densities at 0- to 1-m and 0- to 20-cm depths (Fig. 4). However, the percentage of SOC in the top 0 to 20 cm differed between metropolitan areas with Baltimore having significantly (P=0.045) higher surface proportions than in Denver (Fig. 4). Likewise, native SOC densities at both depths for both metropolitan areas did not statistically differ, while rural forest soils in the Baltimore region had 60% higher SOC in the 0 to 20 cm depth than Denver shortgrass steppe soils (P<0.001) (Fig. 5). By contrast, if the SOC densities of remnant forest soils are considered representative of native soils in the Baltimore metropolitan area, then the native soils of Baltimore would be almost 2-fold higher than the native steppe grass soils of Denver (Figs. 1, 2 and 5).

In the post-urban landscapes, the relatively high SOC densities found for turf grass soils were diminished by the lower densities of soils beneath impervious surfaces (Table 2). For example, the overall weighted mean of SOC densities for the Baltimore and Denver metropolitan landscapes was 7.11 and 8.47 kg m^{-2}, respectively, while turf grass soils were 11.0±0.9 kg m^{-2} and 12.7±1.3 kg m^{-2}, respectively (Table 2 and Fig. 4). Using the urban and suburban remnant forest soils (11.0 kg C m^{-2}) as a native soil comparison to the weighted mean of the post-urban Baltimore, we found a net loss of approximately 35.4% inorganic C (Table 2). By contrast, the weighted mean SOC density of the Denver metropolitan area (8.47 kg m^{-2}) was slightly higher than arid shortgrass steppe soils (7.3± 0.53 kg m^{-2}). A similar result would occur if the NRCS database is used to derive native SOC densities for Maryland rural forests (Table 2).

Eight pedons were found in the NRCS database for cultivated soils of the Piedmont in Maryland. The mean SOC density for these soils (5.44 kg m^{-2}) was 50.5 and 23.5% lower than the pre-agricultural and post-urban landscapes, respectively (Table 2). Data for cultivated soils in the Denver region were not available in the NRCS database; however, literature values for cultivated soils in the Colorado Front Range averaged 5.3 kg m^{-2} (Table 1).

Clearly our comparison of SOC densities between the Baltimore and Denver metropolitan areas shows that residential turf grass soils have the capacity to accumulate

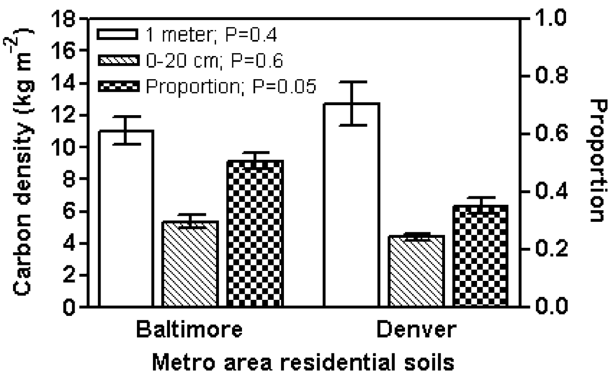

Fig. 4 Mean (±SE) SOC densities (1-m and 0- to 20-cm depths) and the proportion of SOC in the 0- to 20-cm depth of residential soils in Baltimore (n=26) and Denver (n=13) metropolitan areas. Residential SOC density measurements calculated from undisturbed cores. P values represent comparisons of SOC medians using a Kruskal–Wallis test

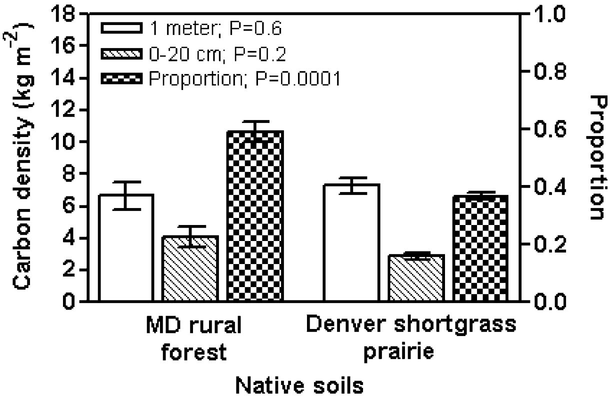

Fig. 5 Mean (±SE) SOC densities (1-m and 0- to 20-cm depths) and the percentage of SOC in the top 0 to 20 cm of native soils in the Baltimore (hardwood forest, n=7) and Denver (shortgrass steppe, n=19 for 1-m and n=35 for 0- to 20-cm depths, respectively) calculated from the NRCS National Soil Characterization database. P values represent comparisons of SOC medians using a Kruskal–Wallis test

large amounts of SOC regardless of regional differences in climate (Figs. 4 and 5). Pouyat et al. (2003) compiled from the literature residential locations reporting SOC densities to a 0- to 1-m depth and calculated a mean SOC density of 15.5±1.2 kg m^{-2} for these areas. Mestdagh et al. (2005) reported a range of SOC between 12.9 and 17.6 kg m^{-2} for grass covered spaces in urban areas of northern Belgium, although none of the locations were described as residential. Kaye et al. (2005) measured SOC densities at a 0- to 30-cm depth in various turf grass areas in Fort Collins, CO, and found more than 45% higher densities in urban than in native shortgrass steppe areas. In the more arid environment of the Phoenix metropolitan area, urban turf grass soils were 44% higher in organic matter concentrations at a 0- to 10-cm depth than in native desert soils (Jenerette et al. 2006).

The most plausible explanation for similarities in SOC densities between turf grass soils in Baltimore and Denver would be the differential management efforts necessary to maintain turf grass in the two climatic regions. Milesi et al. (2005) modeled C and water fluxes of turf grass systems in populated places (>40,000 people) of the conterminous US and found that without irrigation and fertilization, turf grasses would be able to persist in only about 50 of the 865 locations modeled (mostly in the northeastern US). In comparing the regional climates of the Mid-Atlantic and Colorado Front Range, the most significant difference in resource availability is soil water. Indeed, the difference between pan

Table 2 Area-weighted organic C densities of pre-agriculture, cultivated, and post-urban soils for the cities of Baltimore and Denver

	Denver kg m^{-2}-m	Baltimore
Pre-agriculture[a]	7.30	6.7 (11.0)[b]
Agriculture[c]		5.45
Urban[d]	8.47	7.11

[a] Pre-agriculture SOC densities calculated using the NRCS National Soil Characterization database of shortgrass steppe cover types (n=7) in the Colorado Front Range (Denver) and Mid-Atlantic hardwood forest cover types (n=19) in the Piedmont of Maryland (Baltimore)

[b] SOC density in parentheses is for urban and suburban remnant forests

[c] NRCS data for cultivated soils in the Denver region were not available, but were available for Maryland (n=8)

[d] Urban SOC densities calculated using data from Pouyat et al. (2006) and Golubiewski (2006) and include estimated SOC density values for soils beneath impervious surfaces (50.4% and 45% impervious cover for Baltimore and Denver, respectively)

evaporation and precipitation (i.e., evaporative demand) in the shortgrass steppe region of the Colorado Front Range for some years is almost 5-fold higher than that for the Mid-Atlantic states (www.wrcc.dri.edu/climmaps/panevap). This gap in the Colorado Front Range would require high supplements of irrigated water during the growing season to grow turf grass. How much water is required should vary by other site factors such as soil texture; however, it is not clear if and at what point homeowners recognize site factors in making management decisions.

The reason for differences between the percentages of SOC in the top 0- to 20-cm between residential turf grass soils is unclear. We should note that a constant proportion was used to extrapolate SOC densities to a 1-m depth for the Denver cores, which may have introduced an artifact in our computation of the percentage of SOC in the top 0- to 20-cm of these soils. Otherwise, the differences in vertical distribution between the turf grass soils could be due to site history. Jobbágy and Jackson (2000) found significant differences between depths of SOC accumulation among vegetation types with forests having the largest proportion of SOC in surface horizons. These differences were attributed to differences in belowground allocation of biomass among vegetation types, and tree-dominated systems had higher inputs of organic matter than other types at the surface of the soil. Furthermore, various studies have shown that more recalcitrant SOC pools can carry the imprint of previous vegetation for hundreds to thousands of years (cited in Jobbágy and Jackson 2000). Thus, the differences by depth in SOC between turf grass systems of Baltimore and Denver could be explained by differences in the previous cover type (forest vs. grassland) and site history (cropland vs. pasture) of each metropolitan area.

Comparisons in SOC densities between NRCS pedon data of native soil types in both metropolitan regions were not statistically significant (Fig. 5). This was unexpected because SOC densities in the literature suggested that hardwood deciduous forests should have higher densities than arid shortgrass steppe soils (Table 1). Even though NRCS pedon densities did not vary, the proportion of SOC in the 0- to 20-cm depth was higher in the hardwood forest than in the shortgrass steppe soils (Fig. 5). If the comparison is made between native soils of both regions using values obtained for Baltimore from the literature (Table 1) or SOC densities measured in the urban (12.1 kg m^{-2}) and suburban (10.5 kg m^{-2}) remnant forests, the Baltimore native soils would have higher SOC densities than the native soils (shortgrass steppe) in Denver (Fig. 5).

These results suggest that there has been a net reduction in SOC in the Baltimore region (using remnant forests as native soil comparisons) as a result of urban land-use conversion, whereas there has been a slight increase in SOC in Denver, although we were unable to test these differences statistically (Table 2). The difference in the net change in SOC density between metropolitan areas is a result of Baltimore having native soils with higher initial SOC densities and a greater proportion of impervious surfaces than Denver. Although Denver turf grass soils had a higher mean SOC density than those in Baltimore, this difference was not statistically different in the pair wise comparison (Fig. 4). However, if residential turf grass systems in the more arid Denver region prove to have higher accumulations of SOC than Baltimore, the higher accumulations may be due to continual use of automated irrigation systems, which is not likely to occur in the Baltimore region (Law et al. 2004). Other sources of variation in SOC accumulation between turf grass systems may simply be what homeowners do with grass clippings and how often they mow their lawns (e.g., Qian et al. 2003). However, data are not available to evaluate whether differences occurred between Baltimore and Denver areas with respect to these factors.

The results of comparisons made in this analysis generally support our hypothesis that turf grass systems will be similar in SOC densities across regional variations in climate, parent material, topography, and native cover type. These similarities are presumed to be caused by greater management efforts to offset the difference in evaporative demand between the two regions, i.e., anthropogenic factors (management supplements) overwhelmed native soil environmental factors that control SOC accumulation. Thus, in the more arid shortgrass steppe, management efforts to compensate for lower availability of water led to higher accumulations of SOC in turf grass than in the native soils. This finding supports current urban ecological theory, which suggests anthropogenic drivers will overwhelm natural factors in the control of ecosystem processes (Kay et al. 2006).

The hypothesis that native soil types of the same regions should vary due to differences in climate and cover type was not supported by the NRCS soil characterization data for arid shortgrass steppe and Mid-Atlantic hardwood deciduous forests. However, assuming that (1) SOC densities measured for urban and suburban forest remnants of Baltimore (Figs. 1 and 2), and (2) data obtained for hardwood forests from the literature (Table 1) are representative of native forest soils in Maryland, differences of native soils were higher than for residential soils in the regional comparison.

Although managed turf grass systems have the potential to accumulate SOC at relatively high rates, a biogeochemical budget of the carbon cycle relative to turf grass maintenance must be developed to determine actual rates of carbon sequestration (Gordon et al. 1996; Pouyat et al. 2006). Carbon emissions are associated with lawn mowing, energy used to irrigate, and transporting fertilizers (Pataki et al. 2006). Moreover, fertilizers require very high amounts of energy in production (Vitousek et al. 1997). In addition to a budget analysis, more regional and global comparisons of turf grass systems and native soil types of metropolitan areas are needed to fully address the changes occurring in SOC densities due to urban land-use conversion. Specifically, more data are needed to account for intraregional and interregional variations that may occur between different management regimes (Qian et al. 2003; Milesi et al. 2005), intensity of use or disturbance (Pouyat et al. 2002), and site history (e.g., Qian and Follett 2002).

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