Chapter 5
Carbon Stocks in Urban Forest Remnants: Atlanta and Baltimore as Case Studies

Ian D. Yesilonis and Richard V. Pouyat

Abstract Urban environments influence carbon (C) and nitrogen (N) cycles of forest ecosystems by altering plant biomass, litter mass and chemistry, passive and active pools of C and N, and the occurrence and activity of decomposer organisms. It is difficult to determine the net effect of C storage due to the number of environmental factors exerting stress on urban forests. Using a conceptual model to synthesize results from gradient studies of forest patches in metropolitan areas, we attempt to explain the mechanisms affecting C cycling. We also assess the relative importance of C accumulation in urban remnant forests with respect to other land uses previously disturbed or managed. The cities of Baltimore and Atlanta are used as case studies. The C density of forest above-ground biomass for Baltimore City, 8 kg m^{-3}, and Atlanta, 10.6 kg m^{-3}, is significantly higher for both medium- and high-density residential areas. Baltimore City has a forest-soil C density of 10.6 kg m^{-3}, a below-to-above ground ratio of 1.3. Urban forest remnants in these two cities store a high amount of C on a per-unit basis both above- and below ground relative to other land uses, but total C storage is lower due to the lower acreage of urban forest in these cities relative to other land uses.

Keywords Urban • Forest • Carbon • Land use • Management

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5.1 Introduction

Soil carbon (C) pools are roughly three times larger than the C stored in all land plants (Schlesinger and Andrews 2000). At global and continental scales, soil C pools are a function of the inputs of organic matter to the ecosystem from the growth of plants and the loss of C due to decay, both of which are controlled by environmental factors such as temperature and soil moisture. Because of the differences in sensitivities of organic-matter decay and the growth of plants to the environment, there is wide variation in organic soil C densities at continental and global scales (Post et al. 1982). Forests can accumulate C in soil at relatively high amounts compared to other ecosystems due to generally higher inputs of biomass and more recalcitrant organic material (e.g., wood) entering the soil surface (Jobbágy and Jackson 2000).

On a global scale, forest soils constitute about 787 Gt C or about 32% of the total above- and below-ground terrestrial C pool (IPCC 2000). In the conterminous U.S., forest soils in "timberland" hold an estimated 22 Gt of C or 51% of the total above- and below-ground C pools (Heath et al. 2003). By contrast, soil C pools in U.S. urban metropolitan areas are an estimated 1.9 Gt (Pouyat et al. 2006). This approximates the amount of C in forest soils in the Mid-Atlantic States, or 2.27 Gt (Birdsey 1992), and illustrates the importance of urban areas in storing C in the conterminous U.S. (Churkina et al. 2010).

Current research is addressing the question of whether the conversion of natural ecosystems to urban land uses will affect overall storage of terrestrial soil C (Pouyat et al. 2006; Lorenz and Lal 2009). The accuracy of estimates to assess the regional and global effects of urban land-use change on soil C pools is questionable due to uncertainties associated with land-use classification and measurements of C density (Churkina et al. 2010). For forest ecosystems, the long-term effects of forest harvesting are negligible when regrowth of the forest occurs (Johnson and Curtis 2001). By contrast, agricultural uses generally have resulted in greater losses of soil C (Post and Kwon 2000). For urban ecosystems, preliminary evaluations have shown that converting grassland or forest to urban use increase or decrease soil C pools depending on the native climate and soil (Pouyat et al. 2003, 2006). The uncertainty of whether urban landscapes might or might not accumulate more C than the previous native ecosystem, presents an additional challenge in predicting or assessing the effects of land-use change on soil C pools in populated regions of the world.

Forest soils typically have an organic layer overlying a mineral soil horizon that generally is higher in C concentration than in lower horizons of the soil profile. As these soils are converted to urban uses, both direct and indirect factors can affect their pools and fluxes of C (Pouyat et al. 2010). Direct effects include initial disturbances related to urban development that are physical in nature and result in a loss of the organic layer and mineral soil C through oxidation or direct removal of "top soil" for other uses. Post-development, direct effects include the placement in the landscape of impervious surfaces and other physical disturbances that are finer in scale, e.g., trampling and horticultural management (Pouyat et al. 2009b). Direct effects also include the addition of supplements for maximizing horticultural plant and turf grass growth, e.g., fertilizer and water. Indirect effects entail changes in the abiotic and biotic environment as land is converted to urban land uses. This can influence largely undisturbed soils in remnant forest patches, some at distances of tens of kilometers beyond the boundary of an urban area (Pouyat et al. 1995, 2009a). Indirect effects include the urban heat-island effect (Oke 1995; Zhang et al. 2004; Savva et al. 2010), soil hydrophobicity (White and McDonnell 1988; Craul 1992), invasive plant and animal species (Steinberg et al. 1997; Ehrenfeld et al. 2003; Pavao-Zuckerman 2008), and atmospheric deposition of various pollutants (Lovett et al. 2000; Wong et al. 2004). In addition, toxic, sub-lethal, or stress effects of the urban environment on soil decomposers and primary producers can significantly affect decay rates for soil organic matter (Carreiro et al. 1999; Pouyat et al. 2010).

Several authors have suggested that environmental changes that occur with urban land-use are analogous to global climate change such that urbanization gradients can be used to assess these changes on forest C and nitrogen (N) cycling (Pouyat et al. 1995; Carreiro and Tripler 2005). They further suggest that the use of urbanization gradients to assess environmental changes on forest ecosystems constitutes a "whole ecosystem manipulation" that by its very nature is multifactorial and enables the assessment of interactive effects and feedback processes. Here we use these concepts to: (1) review on the potential for urban environmental changes, or indirect effects, to affect the C and N cycles of forest ecosystems, (2) by use of a conceptual model, synthesize results from gradient studies of forest patches in the metropolitan areas of New York City and Baltimore City as well as from other studies, and (3) assess the relative importance of C accumulation in remnant forests with respect to other land-use and cover types that previously were disturbed or historically managed, i.e., direct effects, in urban landscapes of Atlanta and Baltimore City.

5.2 Urban Environmental Effects on Forest C and N Cycles

As mentioned, urban environmental changes can greatly alter the cycling of C and N in forested ecosystems. In particular, the elevation of air temperatures and CO_2 concentrations that occurs in urban metropolitan areas can increase the Net Primary Productivity (NPP) of vegetation or biomass in urban landscapes (Ziska et al. 2004). As with climate change and the C cycle at global scales, the increase in temperature and carbon dioxide (CO_2) and the expected increase in NPP should slow increasing atmospheric levels of CO_2 in urban areas. However, this growth response may persist only for several years as suggested by recent long-term research that shows that plants eventually acclimate to the warmer temperatures and uptake less CO_2 (Hyvonen et al. 2007).

By contrast, the warming due to climate change also is expected to occur asymmetrically as night temperatures, particularly at the low end of the range, are expected to increase more than the maximal temperatures. This nocturnal warming has been postulated to reduce the net C uptake by plants due to greater respiration losses at night. Again, however, studies have shown that many plants overcompensate for the higher respiration rates with higher photosynthesis rates during the day (Wan et al. 2002). As a result, vegetation in urban landscapes may serve as a significant C sink for increasing emissions of CO_2 in urban areas. For example, measurements of eddy flux in Baltimore's metropolitan area suggest a strong vegetation sink for C; however, C sources overwhelm this sink so that there is a net increase in atmospheric CO_2 (Saliendra et al. 2009).

Urban environments also have higher deposition rates of N than surrounding areas (e.g., Lovett et al. 2000; Fenn and Bytnerowicz 1993). Increasing the availability of N to plants generally reduces the use efficiency of N in the plant, resulting in higher concentrations of N in leaf tissue (Throop and Lerdau 2004). However, counteracting the increase in N availability is the effect of elevated atmospheric concentrations of CO_2, which increase N-use efficiencies and thus reduce the concentration of N in leaf tissue (Penulelas and Matamala 1990; Korner and Miglietta 1994; Hyvonen et al. 2007). These effects on tissue concentrations of N will affect both herbivory (Throop and Lerdau 2004) and decay rates in the soil (Hyvonen et al. 2007). Moreover, increases in N availability and atmospheric CO_2 can have significant effects on secondary chemicals in leaf litter, which, in turn, affects soil decay rates.

5.3 Conceptual Model of Net Effects of Urban Environmental Change on C and N Cycles

As occurs at global and regional scales, changes in soil C and N pools in metropolitan areas are dependent on the effects of urban environmental factors (abiotic and biotic) on decay and plant productivity. However, at this higher resolution both the quantity and quality of leaf-litter inputs and the presence, activity, and abundance of decomposing organisms become important factors. Using a conceptual model (Fig. 5.1) based on Carreiro et al. (2009), both the C and N cycles are coupled at metropolitan and regional scales through the availability of N to soil decomposers—the higher the quality of litter (C in Fig. 5.1), the higher the decay of organic matter and thus N availability to plants. In turn, higher N availability to plants produces higher net primary productivity (NPP and F in Fig. 5.1) because N is limiting to growth in most plant communities. However, as more N becomes available to soil microbes the demand for N eventually is exceeded, resulting in excess losses of N from the system, or a state of N saturation (Aber et al. 1998). Soil microbial populations also are limited by daily and seasonal changes in soil moisture and temperature (B in Fig. 5.1).

In forest patches affected by urban environments, it is unclear whether the effect of multiple environmental factors will interact in their effect on plant and decomposer responses (Pouyat et al. 2009b). Several response scenarios are possible given the relationship of the model components in Fig. 5.1. For instance, decay rates can be stimulated by urban environmental factors, thus resulting in higher N availability, which will increase NPP and N concentrations in litter. Another possibility is depression in decay rate and thus reduced N availability and lower concentrations of N in litter (C in Fig. 5.1). Other processes, include plant responses to pollution stress, might lower NPP or reduce litter quality (A in Fig. 5.1); responses of soil organisms to pollution stress, which would slow decay and N transformation rates (B in Fig. 5.1); and the introduction of invasive species, plant or soil invertebrate, that can shift NPP

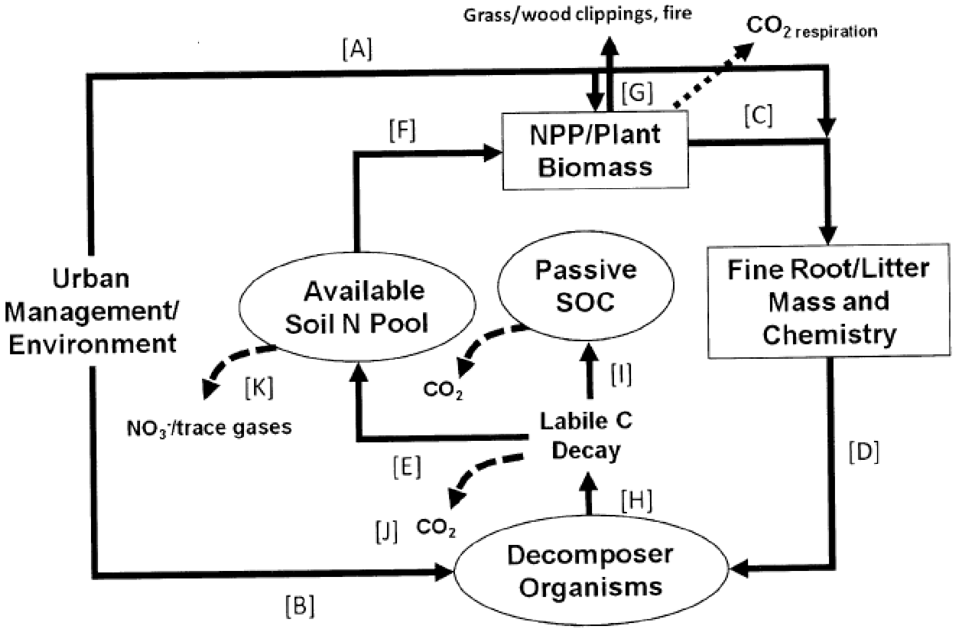

Fig. 5.1 Conceptual diagram of urban environmental change and urban landscape management effects on carbon cycling. Processes are represented as capital letters and explained in the text. *Dashed lines* represent gaseous losses from the ecological system, *solid lines* represent inter connective relationships such as stresses and enrichment between the nutrient pools, flora, and fauna inside boxes. *Rectangles* represent plant pools, *ovals* represent soil pools, and those *not enclosed in boxes* represent broader effects on the system (Modified from Carreiro et al. 2009)

or decay rates of the entire ecosystem (A in Fig. 5.1). Also, the potential for feedbacks has been predicted as well, for examples, increases or decreases in fine-root production in response to higher concentrations of N availability (Nadelhoffer 2000), N inhibition of the production of lignin digesting enzymes (Carreiro et al. 2000), or increases in C or lignin to N ratios in leaf litter produced under higher atmospheric concentrations of CO_2 (Hyvonen et al. 2007) (D in Fig. 5.1).

The net effect of these interactions as shown in Fig. 5.1 can result in either an increase or a decrease in soil C pools. For example, urban environments have been shown to both decrease (Kjelgren and Clark 1992) and increase (Gregg et al. 2003; Ziska et al. 2003) the growth of plants, and increase NPP of entire plant assemblages in one case (Ziska et al. 2004). All other factors being equal, greater inputs into soil of organic matter should result in a higher accumulation of C in forest soils. However, as mentioned previously, urban environments (elevated CO_2 concentrations and increased N inputs) can affect leaf-litter quality, which, in turn, would alter the decay rate of the incoming organic material. Carreiro et al. (1999) and Pouyat and Carreiro (2003) found that red oak (*Quercus rubra*) leaf litter derived from urban forest remnants was of lower quality than rural-derived litter. These differences were reflected in decay rates of the two litters. These are preliminary results as there may be differences in litter quality across species, from year to year, and across urbanization gradients.

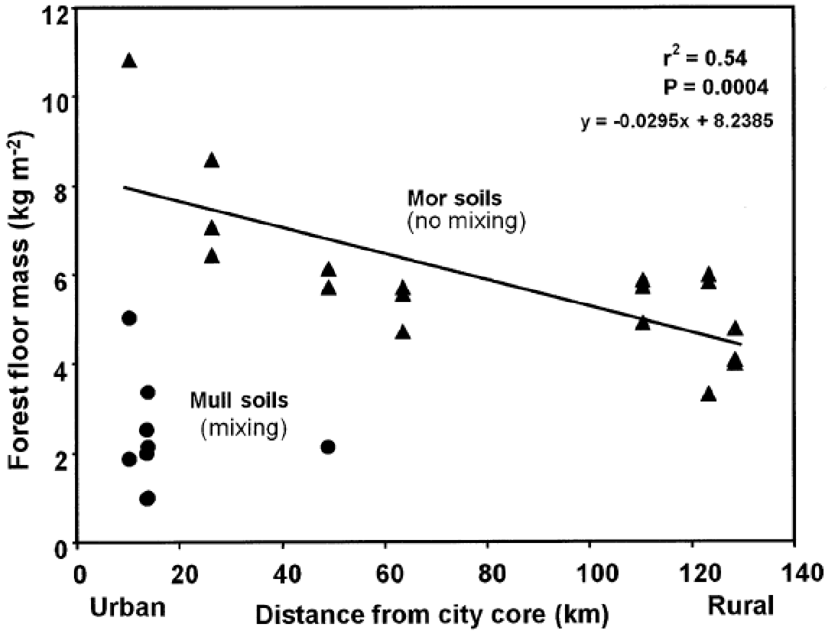

Fig. 5.2 Forest-floor mass of a New York City urban-rural gradient comparing mor soils (*triangles*: no mixing of the O and A horizon) to mull soils (*circles*: mixing of O and A horizon) (Modified from Pouyat et al. 2002)

The conceptual model also emphasizes the influence of urban environmental changes on the soil community (B in Fig. 5.1). As mentioned, the introduction of plant and soil invertebrate species greatly alters C and N cycles in terrestrial ecosystems. For many invasive plant species with relatively low N-use efficiencies, the N concentration of litter inputs can be greatly increased (Ehrenfeld 2003). Introduced peregrine species of earthworms may benefit from urban environmental changes, e.g., longer growing seasons, and, in the short term, greatly accelerate litter decay and N transformation rates (Pavao-Zuckerman 2008; Szlavecz et al. 2006). Over the long term, there are changes in the fungal bacterial ratios of the soil microbial community (Bardgett et al. 1999).

Therefore, the net effect of urban environmental changes on C pools and fluxes in forest soils will be the interaction of those factors impacting NPP and litter quality with those factors affecting the decay of organic matter in the soil (Fig. 5.1). The C response in remnant forest soils along urbanization gradients has been inconclusive. In the New York metropolitan area, evidence suggests little difference in total soil organic C stocks across an urbanization gradient, though the distribution of C in the profile differs (Pouyat et al. 2002). Moreover, the pools of C, as determined by their turnover times, varied along the gradient as urban forest soils had larger recalcitrant pools than rural soils (Groffman et al. 1995). However, C that accumulated in the organic horizons varied due to earthworm activity. For forest patches without earthworms, i.e. mor soils, the forest-floor mass in relation to distance to the urban core was related indirectly (solid triangles in Fig. 5.2). If patches with earthworms are included, i.e. mull soils, the relationship is reversed so that forest-floor mass is directly related to distance from the urban core (open triangles in Fig. 5.2). The net result suggests that litter inputs of poorer quality are compensated for by the presence and activity of invasive earthworm species, without which the soil system would accumulate more C due to poorer quality inputs (Pouyat and Carreiro 2003).

Trace gas fluxes from remnant forest patches also are affected by urban environmental changes. In the Baltimore metropolitan area, long-term monitoring of urban and rural forest patches have shown that urban patches have significantly higher CO_2 fluxes than their rural counterparts (Groffman et al. 2006). By contrast, methane consumption by these soils was significantly higher in rural than in urban forest patches (Groffman and Pouyat 2009). This is consistent with measurements of methane flux in forest soils along an urban-rural gradient in the New York metropolitan area (Goldman et al. 1995). These differences were attributed to N inputs by atmospheric deposition. However, the mechanisms behind these responses are unclear, necessitating the need for controlled, multifactoral experiments, or for field manipulations using treatment ranges determined by observations along the gradient (Pouyat et al. 2009b). Other studies of urbanization gradients have shown that in smaller cities and metropolitan areas with a more compact development pattern, the factors measured in the larger New York City and Baltimore metropolitan areas sometimes differed in importance (e.g., Carreiro et al. 2009; Pouyat et al. 2009b). Thus, the net effect of urban land-use change on C and N cycles of remnant forest patches apparently, depends on the differential effects of urban environmental change on both decay rates and the quantity and quality of leaf-litter inputs into the soil decomposer system, all of which may vary from city to city (Fig. 5.1).

5.4 Direct Effects

As mentioned previously, when land is converted to urban uses, the initial and post-development factors that physically disrupt soil or result from horticultural management, e.g., fertilization and irrigation, can have profound effects on soil characteristics, including the cycling of C and N (Pouyat et al. 2007; Lorenz and Lal 2009). The spatial pattern of these effects is largely the result of parcelization, or the subdivision of land by ownership, as landscapes are developed for human settlement (Pouyat et al. 2003). The parcelization of the landscape creates distinct parcels or patches with characteristic disturbance and management regimes that will affect soil C and N dynamics. The net result is a mosaic of soil patches that varies in size and configuration depending on human population densities, development patterns, and other factors. The soil mosaic includes also remnants of native systems, e.g. a forest remnant, and together with managed and disturbed patches can be used as a suite of "natural experiments" to study the impact of soil disturbance and management on soil C and N (Pouyat et al. 2009b).

It is generally thought that a conversion of native soil types to urban uses results in losses of C; however, depending on the prevailing climate and native soil types,

C can accumulate in soils of urban landscapes to a level that is greater than that
the native soil replaced (Pouyat et al. 2006, 2009a). The primary cause for increasi
C storage in what were once disturbed soils, is the addition of water and nutrien
which under native soil and climate conditions could otherwise be severely limiti
NPP. Therefore, in climates supporting low NPP, the input of plant biomass f
decomposer organisms also is resulting in low soil C pools (C in Fig. 5.1). With t
introduction of supplements such as irrigated water and fertilizer, NPP increas
more than decay rates with the potential to increase the soil C pool. This increase
even greater if a portion of the NPP is not removed from the system by practic
such as mowing, raking of leaves, and pruning of dead and live tree branches (Now
and Crane 2002). Therefore, any management practice that returns a portion of N
to the soil system should result in greater accumulations of C in urban soils. T
importance of management practices in accumulating C in urban soils suggests th
from parcel to parcel, the variability in soil C pools is strongly dependent on t
management practices of the land owner (Pouyat et al. 2007).

5.4.1 *C and N Cycling in Turf Grass vs. Forest Remnants*

An important characteristic of urban land-use change with respect to C and N cycl
is the replacement of native cover types with lawn cover (Kaye et al. 2005; Mil
et al. 2005; Golubiewski 2006; Pouyat et al. 2009b). The estimated amount of la
cover for the conterminous U.S. is 163,800 ± 35,850 km^2, or 73% of all irrigated c
tivated lands (excluding lawn cover) (Lubowski et al. 2006). With respect to turf-gra
cover, it is estimated that roughly half of all residences apply fertilizers (Law et
2004; Osmond and Hardy 2004), some apply fertilizer at rates similar to or exceedi
those of cropland systems, e.g., > 200 kg ha^{-1} $year^{-1}$ (e.g., Morton et al. 1988).

Although the potential for losses of C and N in urban landscapes can be hig
urban soils can accumulate a surprising amount of C and N compared to agricultu
or native soils (Fig. 5.1). Horticultural management, e.g., fertilization and irrigatio
tend to maximize plant productivity and accumulation of soil organic matter for
given climate or soil type and thus increase the capacity of these soils to store C a
N. This is particularly true of turf-grass systems in which soils are not cultivat
regularly and turf-grass species typically grow through an extended growing seas
relative to most native grassland, forest, and crop ecosystems (Pouyat et al. 200
Groffman et al. 2009). The total C budget for maintaining lawns should include emi
sions resulting from mowing, irrigating, and fertilizing lawns (Pataki et al. 2006).

The literature suggests that turf-grass systems can accumulate C to levels that a
comparable to or exceed other grassland and forested systems. Qian et al. (200
used the CENTURY model to show that N fertilization coupled with replacement
grass clippings increased soil C by as much as 59% compared with sites that we
not fertilized and where clippings were removed. Likewise, in measuring soil
stocks in residential areas in the short-grass prairie of the Colorado Front Rang
Golubiewski (2006) found that soil C in turf-grass areas exceeded that in semi-ar

steppe soil in 40 years. The effects of supplements of fertilizer and water on soil C accumulation exceeded those of other soil-forming factors, e.g., elevation and soil texture. In measuring C sequestration rates in turf-grass soils using C^{14} analysis, Qian et al. (2010) found rates of accumulation between 0.32 and 0.78 Mg ha^{-1} $year^{-1}$ during the first 4 years after turf establishment. These rates are similar in range to 0.9–1.0 Mg ha^{-1} $year^{-1}$ during the first 25 years of lawn establishment using the CENTURY model (Bandaranayake et al. 2003).

Substantiating the ability of turf grasses to accumulate C in urban landscapes, Pouyat et al. (2006) reviewed the literature and found that across different climate and soil types, older residential lawns had surprisingly similar C densities of 14.4 ± 1.2 kg m^{-2} at a depth of 1 m. The authors also compared soil C data for urban and the native soils they replaced and found that remnant patches of native vegetation accounted for as much as 34% of a city's stock of soil C. However, when soils that were "sealed" by impervious surfaces were excluded from the computation, estimated soil C densities rose substantially for the urban land-use and cover types, an indication that soils that were not sealed were able to sequester large amounts of C. The relationship of urban soil C stocks with their native soil counterparts varied depending on the prevailing climate and the nature of the native soil. For example, when the authors compared pre- and post-urban estimates of soil C stocks in six cities, they found the potential for large decreases in soil C post-urban development for cities in the northeastern U.S., where native soils have high soil C densities. In drier climates with low soil C densities, cities tended to have slightly higher soil C densities than the native soil types.

Although turf-grass systems have shown a surprisingly high capacity to sequester C, flux rates of C from these systems appear to be higher than the native systems replaced. For instance, measurements in permanent forest and lawn plots in the Baltimore metropolitan area indicate that fluxes from turf-grass plots generally were higher than at forested sites (Groffman et al. 2009). Other soil-atmosphere exchanges of greenhouse gases, especially nitrous oxide and methane, also have been altered by turf management. For example, trace-gas measurements in the Baltimore metropolitan area showed that turf-grass soils have a reduced rate of methane uptake and increased nitrous oxide fluxes compared to rural forest soils (Groffman and Pouyat 2009; Groffman et al. 2009). Similarly, in the Colorado Front Range, turf-grass systems had reduced methane uptake and increased nitrous oxide fluxes relative to native short-grass steppe in that region (Kaye et al. 2004). As with the forest-remnant comparisons along urban-rural gradients discussed earlier, the specific mechanism for elevated CO_2 and nitrous oxide fluxes and a reduced methane sink in turf-grass systems has not been determined, though it would seem that higher atmospheric concentrations of CO_2, N inputs from fertilization, and elevated atmospheric and soil temperatures play significant roles in these soil-flux responses (J and K in Fig. 5.1).

An interesting comparison of soil C and N cycle responses along urban-rural gradients with those in turf-grass systems concerns the difference in the magnitude of the response to management (inferred from land-use type) and urban environmental changes. In measuring potential N mineralization and nitrification rates among forested, turf-grass, and agricultural plots, Groffman et al. (2009) found large differences among the land-use types. When these data were compared to data collected in forest soils along an urban-rural gradient in the Baltimore metropolitan area (Szlavecz et al. 2006), there was as much as a 50-fold difference between the land-use types vs. a 10-fold difference between urban and rural forest remnants (Pouyat et al. 2009a). These results suggest that soil management associated with different land uses has a much greater effect on N cycling than urban environmental changes along urban-rural gradients (Pouyat et al. 2010).

In the following sections we discuss the net effect of urban land-use change on above-and below-ground C stocks by comparing various land-use and cover types Atlanta and Baltimore. These comparisons do not include emissions as a result of maintaining lawns, which could negate the accumulation of C in turf-grass soils (Pataki et al. 2006).

5.4.2 Net Effects of Urban Land-Use Change on C Stocks

5.4.2.1 Comparison of C Sequestration and Stocks of Forest Remnants and Other Urban Land Uses

We present the net effect of urban land-use change on C stocks at the city and state scales. Using data from Atlanta and Baltimore to compare total soil C stocks of forest remnants with those of other urban land-use and cover types. These cities are similar in climate and dominant soil order, but differ in tree canopy cover.

Situated in the Southern Piedmont, Atlanta is dominated by Ultisols, Inceptisols, and Alfisols (USDA NRCS 2006). These same soils are found in Baltimore's Northern Piedmont, which approximates half of that city's area. The other half falls in the Northern Coastal Plain, which is dominated by Ultisols. Both cities fall in the udic soil moisture regime but Atlanta is hotter, falling in the thermic soil temperature regime versus Baltimore's mesic regime. Both cities have kaolinitic or mixed mineralogy, and, their soils generally are loamy. Atlanta receives 20.3 cm more annual rainfall (total of 127.5 cm). The average temperature is 3°C hotter (high average: 22.2°C) and 1.3°C warmer (low average: 11.28°C).

Atlanta is 1.5 times larger in area than Baltimore (341.4–209.2 km^2). Its forest cover is 1.5 times greater in area, or 13.1 compared to 8.1% of the land base. The urban tree canopy cover in Atlanta is 41.7% by land area versus 28% in Baltimore. Atlanta and Baltimore have 752 and 598 trees/ha, respectively, well above the mean of 409 for two other comparable U.S. cities: Philadelphia, PA and Washington, DC (Nowak and Greenfield 2009).

There are few data on the tree and soil C content of urban forest remnants. We used forest remnant data from the I-Tree Eco model, whose urban land use classifications includes "forest" (Nowak and Greenfield 2009). I-Tree Eco is a model to assess urban forest tree canopies through field collection of tree data stratified by land use. Atlanta and Baltimore were the only cities for which forest was a specified land use. Other cities had broader classifications, which would increase the variation

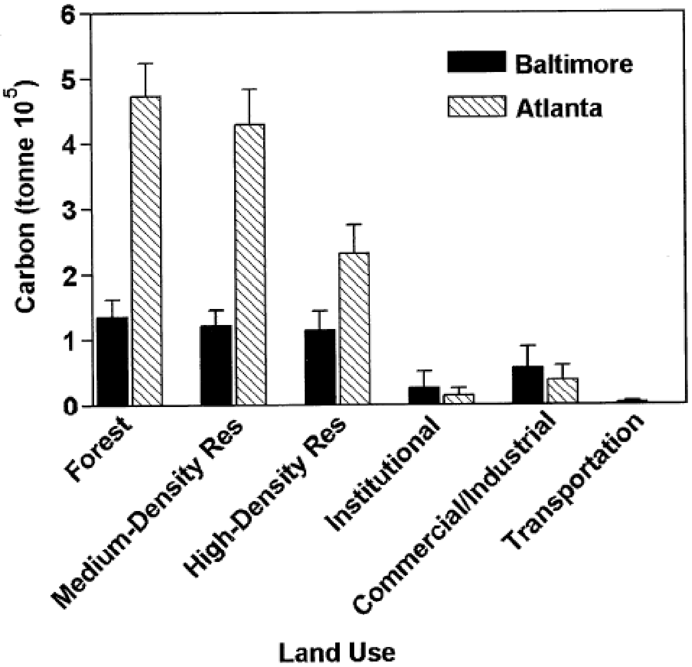

Fig. 5.3 Mean (S.E.) of total carbon storage (10^5 tonnes) in trees for *Baltimore* and *Atlanta* by land use (Data source: Nowak and Greenfield 2009)

of the measured metrics and would be less specific to urban forest, for example, Chicago, for which the "vacant" land-use designation included measurements of forest-land use, Oakland (miscellaneous), Boston (urban open), and Syracuse and San Francisco (green space and vacant). Atlanta was chosen for comparison with Baltimore because of a specific land use: forests.

Northeastern forests (8.23 kg m^{-3}, Heath et al 2003) are about 1.8 kg m^{-3} higher in C stored in live trees and shrubs than Southeastern forests (6.39 kg m^{-3}, Heath et al 2003). Therefore, in urban environments, we would expect similar values relative to each other, i.e., a city in the Northeast (Baltimore) should be approximately 1.8 kg m^{-3} greater in above-round C than a city in the Southeast (Atlanta). However, Baltimore is 3.3% lower than the regional forest-stand estimate while Atlanta is 65% higher. This leads us to believe that urban forest structure varies considerably between Atlanta and Baltimore. Additional research is needed to determine why this occurs.

The importance of forest remnants in the overall breakdown of C stocks for urban landscapes is evident. Total C storage in trees in Baltimore is about equal in the land uses of forest, medium- and high-density residential (about 130,000 tonnes) (Fig. 5.3). In Atlanta, forest and medium-density residential areas store about 450,000 tonnes of C, followed by a 50% reduction of C in high-density residential use (Fig. 5.3). Both cities contain a small percentage of total C storage of trees in the institutional, commercial-industrial, and transportation land uses. These results are driven more by land-use area than by the C density, for example, Baltimore has

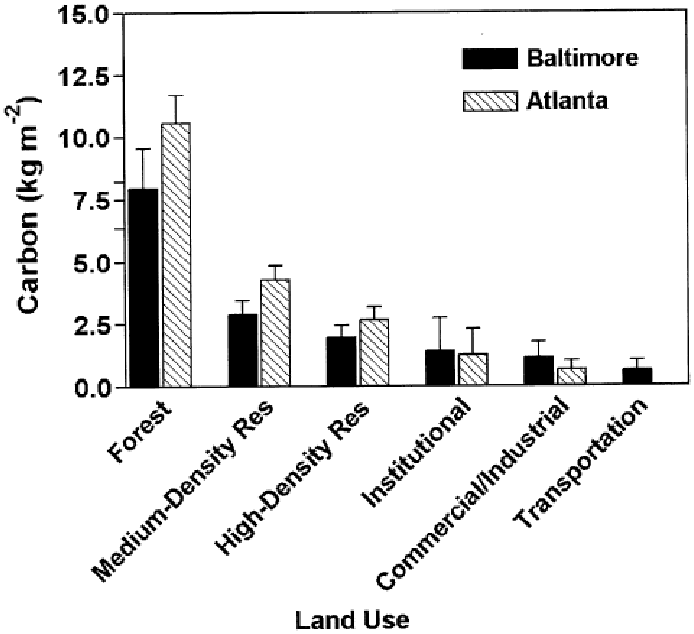

Fig. 5.4 Mean (S.E.) of carbon density (kg m^{-2}) of trees for *Baltimore* and *Atlanta* by land use (Data source: Nowak and Greenfield 2009)

40% more medium-density residential area compared to forest (1,698, 4,192, and 5,870 ha for forest, medium- and high- density residential uses, respectively). Therefore, it is important to examine C density, which is normalized C content by area.

The C density of above-ground biomass of forests in Baltimore and Atlanta is significantly higher than for both medium- and high-density residential uses (Fig. 5.4). The forest is 8 kg m^{-3} in Baltimore and 10.6 kg m^{-3} in Atlanta (these figures include both pervious and impervious areas). Forest exceeds the residential areas by 5.6 and 7.1 kg m^{-3} for Baltimore and Atlanta, respectively. When normalized, the impact of forested systems on overall soil C stocks is apparent (Fig. 5.4).

Soils constitute a major component of the total storage comparison, which is evident in Baltimore with a below-to-above-ground C ratio of 1.3. This is consistent with ratios for other ecosystems, e.g., rural forest land in the Northeast (1.14) and Southeast (1.20) (Heath et al. 2003). For the soils in Baltimore (data for Atlanta soils not available), we would expect less C in remnant forest soils in cities than in forest stands in the Northeast (13.40 kg m^{-3}, Heath et al 2003) due to trampling or a previous disturbance. Baltimore's remnant forest soils have 20% less C (10.6 kg m^{-3}) (Fig. 5.5), than rural forest soils in the Northeast. However, compared to Maryland's rural forests (7.7 kg m^{-3}), there was a 38% increase of C in urban versus rural soils (Pouyat et al. 2009a). This may be related to extensive agricultural practices that began early in the seventeenth century and continued for nearly two centuries. Urban forest remnants may be older than rural forests because agricultural abandonment may have occurred earlier in populated areas (Pouyat et al. 2009a).

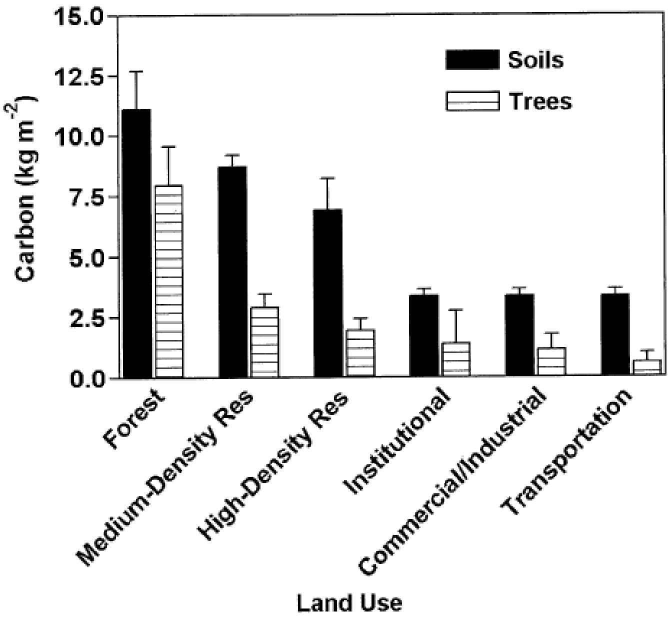

Fig. 5.5 Mean (S.E.) of carbon density (kg m^{-2}) (both pervious and impervious) of *soils* and *trees* in Baltimore by land use (Data source: Nowak and Greenfield 2009; Pouyat and Yesilonis, unpublished)

On the basis of previous analysis, soil in Baltimore apparently accounts for most of C stored in the city environment relative to trees. To estimate total storage of soil C in Baltimore, we sampled 24 plots to a depth of 1 m within forest and residential land uses and assumed 3.3 kg m^{-3} C density below impervious surfaces (Pouyat et al 2006) medium- and high-density residential areas account for most of the soil C stored in Baltimore followed by similar values in forest and commercial-industrial land uses (Fig. 5.6). As with above-ground biomass, land-use area accounts for these differences.

There is a difference in C storage between pervious and impervious cover. The difference in soil C, is small for forest and medium-density residential; the impervious areas have a more significant effect for other land uses. Since there is only 12% impervious area in the forest, the difference in soil C between the impervious + pervious and pervious alone is negligible. Medium-density residential contains 41% impervious area versus 64% for high-density residential; other land uses have more impervious surfaces. The importance of understanding the difference between pervious and impervious is reflected in the totals derived from six cities in the I-Tree Eco analysis by Pouyat et al. (2006): Atlanta, Baltimore, Boston, Chicago, Oakland, and Syracuse. (The soil data used for this analysis was compiled from various sources for each city.) Soils, excluding those under impervious cover, have a C density of 9.5 kg m^{-3} (Pouyat et al 2006); the decrease is significant when impervious areas are included (average C density of 6.3 kg m^{-3}). The above-ground C density is 4.4 and 2.1 kg m^{-3} when impervious areas are included.

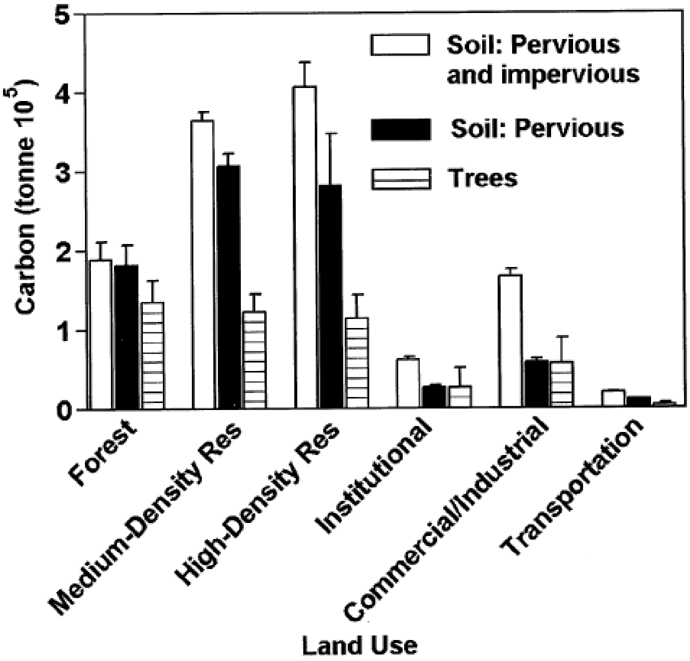

Fig. 5.6 Mean (S.E.) of total carbon storage of trees and soils (10^5 tonnes) for Baltimore by land use. Soils are divided by those under pervious surfaces only, i.e., no impermeable layer above the surface horizon, and those under pervious and impervious surfaces. The latter include soils under pervious surfaces and those under impermeable layers such as buildings and streets (Data source: Nowak and Greenfield 2009; Pouyat and Yesilonis, unpublished)

5.5 Conclusions

Urban land-use changes, indirect and direct, have significant and quantifiable effects on forest C in urban areas. A framework to investigate these controlling factors (Fig. 5.1) in forest remnants shows how the urban environment and management affect C cycling and thus increase or decrease C storage. Forest remnants can support a considerable amount of C storage even in highly urbanized landscapes (up to 20% of total C stored in Baltimore). Tree C densities in Baltimore and Atlanta are similar, though there is a great difference in total C storage that favors highly treed Atlanta. Most of the soil C in Baltimore is associated with land uses covering large areas, e.g., residential, even though densities are similar between forest and residential land uses. Land-use effects have a greater effect on C storage than environmental factors. A difference between forest and residential land uses is associated with the financial and environmental cost of management, i.e. fertilizers and nutrient flux. It also was determined that the net effect of urban land-use change is dependent on the proportion of landscape disturbed, or left remnant, and the proportion left under impervious surfaces.

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