

# Chapter 7

## Carbon Storage in Some Urban Forest Soils of Columbus, Ohio, USA

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**Abstract** Soils store more carbon (C) in soil organic matter (SOM) and in carbonates than the vegetation C and atmospheric carbon dioxide (CO<sub>2</sub>)-C combined. Specifically, forest soils are a major C store. For example, about 8% of the global soil C is stored in soils of temperate forests to 3-m depth. However, data on soil C storage in urban forests are scanty. In the U.S., about 10% of the terrestrial C storage is located in human settlements, of which 64% is stored in soils. Further, soils under urban forests in the U.S. store about three-times as much C to 1-m depth as is stored in the tree biomass. In Ohio, about 35 megagram (1 Mg=10<sup>6</sup> g) C ha<sup>-1</sup> are stored in urban trees but there are few if any available reports on urban forest soil C storage. Ohio is a rapidly urbanizing state and farmland is increasingly converted into urban land uses. However, urbanization is also accompanied by planting of trees and the establishment of urban forests. Thus, a study was conducted to assess soil C storage to 1-m depth in two urban forests in Columbus, Ohio: Clinton-Como Park (CP) and Driving Park (DP). Both forests were disturbed by recreational activities. In addition, CP sited at the east bank of the Olentangy river is also disturbed by flooding and prior levee construction activities. The DP is sited on a former race track, and is also disturbed by previous railway dam construction activities and municipal solid waste disposal. Ten soil samples were randomly obtained per site, and analyzed for bulk density, and total C and nitrogen (N) concentrations for

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computation of the soil C pool. Differences in C concentrations to 1-m depth among site were small but higher to 30-cm depth than those reported under urban tree cover in Colorado, USA ( $>1.60\%$  C). Further, N concentrations in sub-soil horizons were higher at CP than at DP ( $0.13\%$  N vs.  $0.7\text{--}0.9\%$  N below 70-cm depth at CP and DP, respectively). Soil N concentrations in upper soil horizons at both urban forests were higher than those in Baltimore, MD, and Colorado. Similar amounts of C were stored in both soils in Columbus to 1-m depth ( $211$  and  $163$  Mg Cha<sup>-1</sup> at CP and DP, respectively). Thus, soil C pools in urban forests were higher than those to 1-m depth reported for wooded areas in New York City, NY ( $97\text{--}145$  Mg Cha<sup>-1</sup>). However, the soils from Columbus must also to be analyzed for inorganic C and coal C to estimate the net soil organic carbon (SOC) pool. Estimates of the net SOC pool will allow the comparison with estimates for non-urban forests soils in Ohio and urban soils in the U.S., and of the assessment of the C sequestration potential in soils under urban ecosystems in Ohio.

**Keywords** Urban forest soils • Soil carbon pool • Urban land uses • Columbus • Ohio

## 7.1 Introduction

Forest ecosystems occupy the largest part of ice-free land surface among all terrestrial ecosystems. Annually, forests absorb large amounts of carbon dioxide (CO<sub>2</sub>) from the atmosphere via photosynthesis, and return a large part of the fixed carbon (C) back to the atmosphere through auto- and heterotrophic respirations. However, a small fraction of assimilated C is stored in above- and belowground biomass, litter, and soil. Thus, global forests are sinks for atmospheric CO<sub>2</sub> by taking up between  $1.58$  and  $2.37$  petagram ( $1$  Pg =  $10^{15}$  g) C year<sup>-1</sup> (Lorenz and Lal 2010). This rate of C uptake corresponds to about half of the terrestrial C sink (Canadell et al. 2007). Further, forests store higher amounts of C than is stored in the atmospheric carbon dioxide (CO<sub>2</sub>) pool. Specifically, it has been estimated that between  $357$  and  $691$  Pg C are stored in trees, and between  $705$  and  $968$  Pg C may be stored in soils beneath the global forest to 1-m depth (Lorenz and Lal 2010). About  $8\%$  of the global soil C pool to 3-m depth is stored in soils beneath temperate forests ( $262$  Pg C vs.  $3,200$  Pg C; Jobbágy and Jackson 2000; Sundquist et al. 2009). In particular, pristine, undisturbed forests sequester C and are important components of the terrestrial C cycle by slowing-down anthropogenic increases in atmospheric CO<sub>2</sub> caused by fossil fuel burning and land use changes.

Forest ecosystems are directly affected by urbanization (i.e., the expansion of urban land uses, including commercial, industrial, and residential uses). Urbanization is one of the most dramatic and dynamic global human alteration of ecosystems (Grimm et al. 2008). The adverse environmental effects of urbanization are exacerbated by the current extremely rapid rate of urbanization which is unprecedented in human history (UNFPA 2007). In particular, the conversion of vegetated land such as forests by deforestation to urban land causes a reduction in the net ecosystem exchange (NEE) or net CO<sub>2</sub> exchange, and possibly also the net ecosystem C balance

(NECB) (Chapin et al. 2006; Trusilova and Churkina 2008). The NECB is the forest C balance from all sources and sinks. In addition, construction of homes and associated structures among forests at the wildland-urban interface potentially reduces the forest C sink (Radeloff et al. 2005). However, urban ecosystems are also the major source of CO<sub>2</sub> emissions contributing to greenhouse gas (GHG) forced climate change (IEA 2008). In addition to climate change, cities are also exposed to localized climate effects of urbanization such as the urban heat island (McCarthy et al. 2010). However, urbanization effects are seldom included in coupled biosphere-atmosphere models to simulate the abrupt climate change (ACC), for example, by coupling an urban land-surface model to a global climate model (Bonan 2008). Through a simulation study, McCarthy et al. (2010) concluded that climate change has the capacity to modify the climatic potential for urban heat islands, with increases of 30% in some locations, but a global average reduction of 6%. Further, warming and extreme heat events due to urbanization and increased energy consumption were simulated to be as large as the impact of doubling of CO<sub>2</sub> in some regions, and climate change may increase the disparity in extreme hot nights between rural and urban ecosystems (McCarthy et al. 2010).

There are few if any published reports on assessments of the SOC pool and its dynamic in urban ecosystems (Lorenz and Lal 2009; Rawlins et al. 2008). The science of the C cycle and measurements of C pools and fluxes has mostly focused on natural ecosystems, and ignored urban settlements (Pataki 2007; Lorenz and Lal 2009). However, urban ecosystems may contribute to terrestrial C sequestration as they may include protected forests, unprotected (or undeveloped) forest areas, and trees grown around a house or in the neighborhood surrounding the house (Mansfield et al. 2005). Thus, urban forests and trees may directly contribute to mitigating ACC through C sequestration in biomass, litter and soil, and affect the urban climate through biophysical effects. Increasing the amount of ‘green space’ infrastructure such as trees and reducing the amount of impervious ‘gray’ infrastructure (e.g., buildings and roads) has therefore been proposed to reduce the environmental impact of cities and increase ecosystem services in urban ecosystems (Carreiro 2008).

This Chapter begins with an introduction about the importance of forests in the global C cycle, and the effects of urbanization on the functions of urban forests with a focus on studies from the United States. Then, results for two urban forest soils in Columbus are presented. Finally, an outlook is given about studies needed to address the importance of urban forests for the regional C balance.

## 7.2 Urban Trees and Forests

### 7.2.1 Definitions

A ‘forest’ is defined as a land spanning more than 0.5 hectare (ha) with trees taller than 5 m and a canopy cover of more than 10%, or trees able to reach these thresholds in situ (FAO 2006). A forest ecosystem is defined both by the presence of trees

and the absence of other land uses. Urban forests, in particular, can be distinguished by their location in or near densely-built urban centers (Konijnendijk 1997). Often, urban forests have a high density of recreational facilities and are rather fragmented in size and ownership compared to non-urban forests. Further, ‘urban forestry’ is also defined as the art, science and technology of managing trees and forest resources in and around urban community ecosystems for the physiological, sociological, economic, and aesthetic benefits which trees provide to the society (Konijnendijk et al. 2006). Based on the tradition of shade tree management, urban forestry has a particularly long history in North America. The definition of urban forestry is now more comprehensive, including all tree stands and individual trees in and around urban areas, and the multifunctional and multidisciplinary character of urban forestry and urban forests is acknowledged (Konijnendijk et al. 2006). Thus, urban forest refers to all woody plants in and around the city, including street trees, yard trees, park trees, and planted or remnant forest stands (Wu 2008).

### ***7.2.2 Functions in the Climate System***

Urban forests have many environmental and economic benefits (Carreiro 2008). Specifically, they improve air quality by absorbing particulates and pollutants (e.g., ozone, chlorine, sulfur dioxide, nitrogen dioxide, fluorine), improve water quality by retaining nutrients and contaminants in SOM, reduce noise pollution, control floods, reduce soil erosion, moderate the urban heat island, reduce the energy required to cool and heat buildings, increase real estate values, and improve the supply of drinking water in urban and in ex-urban ecosystems beyond cities (Wu 2008). Urban trees and forests directly contribute to sequestration of atmospheric CO<sub>2</sub> in urban ecosystems as urban trees fix CO<sub>2</sub> during photosynthesis and store C in biomass, litter and soil (Long and Nair 1999). However, C fluxes through urban forests are neglected in terrestrial C cycle models (Churkina 2008). Also, credible estimates of the fraction of the total area under forest allocated to urban forests are not available (Churkina et al. 2010). In addition to scanty data about SOC, even less is known about the inorganic C storage in soils beneath urban forests.

In addition to C sequestration, urban forests have other important biophysical effects on the ACC (Ryan et al. 2010). Specifically, urban forests directly impact the urban air temperature because these ecosystems differ from other urban surfaces in moisture regime, aerodynamic and thermal properties (Bowler et al. 2010). The cooling effect by urban forests results primarily from evapotranspiration or the loss of water vapour from a tree into the atmosphere. Evapotranspiration uses energy from solar radiation and increases the latent rather than the sensible heat, thereby cooling the leaf and the temperature of the air surrounding it. Further, shading by urban trees cools the atmosphere by intercepting the solar radiation and preventing the warming of the soil surface and air (Oke 1989). For example, parking lot trees in Davis, CA, reduced the surface temperature of asphalt by as much as 20°C, and cabin temperatures of vehicles by over 26°C (USDA/CUFR 2002). Thus, urban

forests can have lower temperatures during the day compared to surrounding non-green urban spaces (Bowler et al. 2010). By lowering temperatures and shading buildings in summer, and by blocking winds in winter, urban trees can partly offset CO<sub>2</sub> emissions from power plants and minimize its radiative forcing (Heisler 1986). Tree planting influences urban air temperatures by altering albedo, shading, and changing the latent heat flux (Pataki et al. 2009). Thus, large benefits of urban tree planting in terms of ACC mitigation are similar to the effects of afforestation on surface energy balance rather than those from direct C sequestration. However, the urban tree forest canopy may retain heat at night making it difficult to assess the net benefits of larger urban forests on ACC (Ryan et al. 2010; Huang et al. 2008).

Urban forestry projects which reduce energy use positively impact the urban climate (Jackson et al. 2008). Promoting tree planting outside of forests, especially in urban ecosystems is an adaptive forest management practice to mitigate ACC (Bravo et al. 2008). The biomass from urban trees and wood by-products can be a source of bio-based fuels for power and heat generation, thereby reducing the fossil fuel consumption (Mead 2005; MacFarlane 2009). Advanced combustion of wood from urban trees, for example, offers environmental benefits, such as renewable energy source in the U.S. (de Richter et al. 2009) and elsewhere. Thus, strategies of ACC mitigation using urban forestry include increasing C density in settlements, using wood from urban trees as renewable energy source, accentuating indirect effects such as reducing the energy use for heating and cooling of buildings, and changing the albedo of paved parking lots and roads (Nabuurs et al. 2007).

Relatively well documented is the C storage of urban trees in the U.S. as urban forestry has a long cultural tradition in this region. On average, forests cover 27% of the urban land area in the continental U.S. of which 3% of the total tree cover comprises of urban ecosystems (Nowak et al. 2001). The C density in aboveground forest vegetation generally ranges between 59.5 and 111.5 Mg Cha<sup>-1</sup> (Nowak and Crane 2002). In total, urban trees in the coterminous U.S. store between 0.36 and 1.0 Pg C, and sequester between 0.014 and 0.026 Pg C year<sup>-1</sup> (Nowak and Crane 2002). However, the C storage in urban trees estimated from the biomass measurements is approximate at best. Specifically, the accuracy of urban tree C biomass estimates is low as allometric relationships developed outside of urban environments cannot be extrapolated to estimate urban forest tree C storage (McHale et al. 2009). The main reason is that urban trees show different growth rates compared to rural trees because of the relatively open structure of urban forests and the proximity to impervious surfaces (Nowak and Crane 2002; Quigley 2004). On unit area basis, however, C storage in rural forests is more than in urban ecosystems which are characterized by less tree coverage. Thus, urban forests in the U.S. have only limited potential to store C compared to non-urban forests, because urban ecosystems occupy a relatively small land area (Ryan et al. 2010). However, assessing the net climate impact of intensively managed urban forests and trees necessitates information on: (i) the C storage rate of the trees, (ii) fossil fuel emissions from energy associated with planting and maintenance, (iii) fossil fuel emissions resulting from the irrigation process, (iv) nitrous and nitric oxide emissions from fertilizer use, and (v) the net effect of forests and trees on local air temperature and its impact on building

energy use. Life cycle analysis (LCA) is needed to assess the net impact. Furthermore, these factors are likely to be highly variable among regions and species (Ryan et al. 2010).

### 7.3 Urbanization and Urban Forests

The current rapid pace of urbanization is unprecedented in human history. Urban lands are the most intensively transformed and drastically perturbed ecosystems. These ecosystems have increased dramatically in area by as much as a factor of 40 between 1700 and 2000 (Ellis et al. 2010). For example, during 1700–2005, urban land in China increased from 1.6 Mha to 18.8 Mha, with the largest increase of 4.9 Mha (+35.7%) between 1980 and 2005 (Liu and Tian 2010). Thus, urbanization is now the primary process of global land cover transformation (Pavao-Zuckerman and Byrne 2009). Urbanization drastically alters the C cycle through land use change, climate modification, and atmospheric pollution (Trusilova and Churkina 2008). However, the global forest area converted to urban land use and its consequences to net primary productivity (NPP) of forests are not well known (DeFries et al. 1999; FAO 2006). In the 1990s, urbanization in Europe caused the loss of about  $0.003 \text{ Pg C yr}^{-1}$  from the terrestrial environment (Zaehle et al. 2007). However, when all urban changes are taken into account, a net increase in C sink was observed. This C sink was attributed to the  $\text{CO}_2$  fertilization effect and nitrogen (N) pollution. But, there are no reliable data in support of this hypothesis (Trusilova and Churkina 2008). Similarly, the net effects of urbanization and density of human settlements on ecosystem C budget in the U.S. are not known (Churkina et al. 2010).

In the U.S., urbanization is occurring at a faster rate than that of the population growth (Pataki 2007). However, assessing the net effect of urbanization on the ecosystem C budget remains a challenge because there are few if any studies on C budget in urban ecosystems. Most of the regions with large urban expansion in the U.S. are heavily forested (Nowak et al. 2005). For example, a spatially detailed analysis indicated that in the short period between 1990 and 2000 the area of developed land in the Chesapeake Bay Watershed increased by 61% of which 33% occurred on forested land (Jantz et al. 2005). Yet, the loss of forest functions may have been much higher because of increased edge effects and fragmentation. Thus, most forests are no longer remote from cities, but are surrounded and penetrated by development and are indirectly affected by urbanization. The increase in developed land area in the Chesapeake Bay region is predicted to consume 14% of forest land in the region by 2030 (Goetz et al. 2004). A total loss of about 16,000  $\text{km}^2$  of forestland to urban development was observed in the U.S. between 1997 and 2001 (Lubowski et al. 2006). Also, much of newly developed land in the U.S. had been forested, i.e., 40% of the land developed in the 1980s, and 46% of the land converted by urban sprawl between 1997 and 2001 (Duryea and Vince 2005). Furthermore, because urbanization occurred in mostly forested areas, such a land

use conversion caused an overall loss in annual NPP in many northern U.S. cities (Imhoff et al. 2004). A majority of new urban and developed land is projected to come from forestland and, thus, the C storage potential of terrestrial ecosystems in the U.S. may be reduced (USGCRP 2003). In particular, significant amounts of U.S. forestland are projected to be transformed by urbanization, i.e., about 5% of forestland outside of urban areas may be directly converted to urban growth between 2000 and 2050 (Nowak and Walton 2005).

## 7.4 Urbanization and Terrestrial Carbon Storage in Ohio

Until mid 1700s, >95% of Ohio was covered by forests (Managing Ohio's forest resources: challenges & opportunities, Ohio Chapter of The Society of American Foresters, <http://www.ohiosaf.org/>). During the subsequent 200 years, almost 85% of the land and, particularly the forest land was converted to row-crop agriculture and other uses (Lafferty 1979). Thus, only 10% of Ohio was covered by forest by 1940s. Since then forest acreage has steadily increased. However, prime agricultural soils remain under agricultural land use (Medley et al. 2003; Simpson et al. 1994). In 1997, 26.8% of Ohio was covered by forestland. By early 2000s, 30% of Ohio, or 3.2 Mha, was forested. The majority of this forestland is located in the eastern and southern unglaciated region of Ohio. More than 300 different tree and shrub species are identified in Ohio, of which at least 20 are among commercially important tree species (Managing Ohio's forest resources: challenges & opportunities, Ohio Chapter of The Society of American Foresters, <http://www.ohiosaf.org/>).

Ohio is among the fastest urbanizing states in the U.S. About 80% of Ohio's citizen live in an urban metropolitan area and this percentage is expected to increase (Urban forestry in Ohio, Ohio Chapter of The Society of American Foresters, <http://www.ohiosaf.org/>). Urban land increased by 22.9% between 1982 and 1997, and 13.7% of Ohio was covered by urban land in 1997 (Irwin and Reece 2002). However, the population growth was modest despite fast rates of urbanization in the 1990s. Thus, low-density development and exurban areas of the state have increased, causing a substantial loss of rural ecosystems. Urbanization rates in Ohio are closely linked to the rate of loss of farmland in and around the metropolitan areas. Between 1992 and 1997, metropolitan areas added urban land at a rate that is more than double the rate of their population growth, leading to a de-concentration of population in Ohio's metropolitan regions. Overall, these trends point to a pattern of urban growth in Ohio that has become increasingly spread out and 'sprawling' over time (Irwin and Reece 2002), leading to changes in forest cover. Specifically, forestland in Ohio increased by 6% between 1982 and 1997, particularly in the east and southeast, mainly because farmland was taken out of production. However, some counties mainly in southeastern Ohio reported also the loss of forest cover. In metropolitan counties, forest cover increased only by 2.9% between 1982 and 1997 but urban land cover in metropolitan counties increased by 28.0% during the same period (Irwin and Reece 2002). During 1999, Ohio municipalities planted 218,643 trees,

pruned or otherwise maintained 381,759 trees, and removed about 69,814 trees (Urban Forestry in Ohio, Ohio Chapter of The Society of American Foresters, <http://www.ohiosaf.org/>). As many as 6,468 tree planting sites are available along the streets of the average Ohio City. However, far too little is known about urban ecosystems in Ohio and the requirements for urban tree survival and growth. Another major adverse environmental impact of urbanization in Ohio is the loss of wetlands (Kaplan et al. 2001). Only 10% of the original wetlands in Ohio still exist. Therefore, the expansion of urbanization and the resulting changes in the landscape is now responsible for major land use changes and the attendant environmental degradation in Ohio (Kaplan et al. 2001).

The land use conversions in Ohio, especially the spread of urban land into rural areas, alter the terrestrial C cycle among having other environmental consequences (Kaplan et al. 2001). However, the potential of the increasing urban land use and management to maximize the C sinks in urban vegetation and soil is neither well understood nor adequately characterized (Pataki et al. 2006). Similar to the Front Range of Colorado, the terrestrial C pools in Ohio may also be underestimated by the exclusion of urban land cover (Golubiewski 2006). For example, Nowak and Crane (2002) estimated that urban areas in Ohio store 0.035 Pg C in aboveground tree biomass with an average density of 35 Mg C ha<sup>-1</sup>. However, the C storage in non-tree biomass (i.e., herbaceous cover and woody plants with diameter at breast height <2.5 cm) was not included in the estimates.

With natural soils having a relatively large soil organic C (SOC) density, post-urban development has the potential of leading to a decrease in the SOC pool (Pouyat et al. 2006). For example, wetlands in Ohio contain large SOC pools that may decrease after conversion into urban land uses and ecosystems (Bernal and Mitsch 2008). In a simulation study using taxonomic and geographic approaches, Tan et al. (2004) estimated that between 0.85 and 0.88 Pg C are stored to 1-m depth in non-urban soils in Ohio with an average density of 102 Mg C ha<sup>-1</sup>. In urban soils of Ohio, an estimated 0.076 Pg C may be stored to 1-m depth with an average density of 81 Mg C ha<sup>-1</sup> (Pouyat et al. 2006). However, these estimates are highly uncertain because these are: (i) not validated against data from urban soils in Ohio, and (ii) many assumptions are required to obtain these estimates. Preliminary studies indicate that 5% and 75% of urban land in Ohio, respectively, is sited on soils of the high and moderately high productivity class (Nizeyimana et al. 2001). Thus, a relatively high proportion of soils suitable for terrestrial C sequestration in Ohio are lost to urban development. Golf courses are an important recreational land use in Ohio. Selhorst (2007) estimated that golf turfgrass soils in central Ohio sequester between 2.64 and 3.55 Mg C ha yr<sup>-1</sup> and store 157.1 Mg C ha<sup>-1</sup> in 0.15-m depth. However, C emissions associated with intensive turfgrass management can render soils into a C source relative to the atmosphere about 30 years after the golf course establishment. Thus, data on terrestrial C storage in urban ecosystems in Ohio are urgently needed to improve estimates of the urban SOC pool, and to predict alterations of the regional C cycle by the rapid urbanization occurring in this region.



## 7.5 Urban Forest Soil Carbon Storage in Columbus

### 7.5.1 *Columbus and Franklin County*

The city of Columbus is located in central Ohio, and sited in the middle of Franklin County at the confluence of the Scioto and Olentangy rivers, with parts of Columbus expanding into both Delaware County and Fairfield County (39°59'00"N; 82°59'00"W). Prior to the arrival of the first European settlers around 1,800, the city area was covered by dense deciduous forests. Forests were cut during the following years to reclaim land for agriculture as the deep, nearly level and gently sloping soils were well suited for farming (SCS 1991). It is likely that the original forest cover in Columbus and Franklin County was removed by clearing and cultivation for agriculture more than 150 years ago (Quigley 2002). The population increased rapidly from 2,435 in 1830 to ~769,360 in 2009 accompanied by an increase in urban land cover (U.S. Bureau of the Census; <http://www.census.gov/>). Between 1982 and 1997, for example, about 42,000 ha urban land (an increase of 31.6% in urban land cover) was added to Columbus primarily due to population growth (Irwin and Reece 2002). Among all the metropolitan areas in Ohio counties, the Columbus metropolitan area recorded the highest percentage increases in urban lands between 1982 and 1992 (Kaplan et al. 2001). In 2010, Columbus covered an area of 550.5 km<sup>2</sup>, and further expansion is occurring mainly on prime farmland (SCS 1991). Between 1974 and 1992, agriculture strongly declined because of decrease in the farmland and conversion to urban uses in the core metropolitan counties of Franklin (Columbus) aside Cuyahoga (Cleveland) and Hamilton (Cincinnati) (Kaplan et al. 2001). Based on the National Land Cover Database (NLCD), about 58% of Franklin County including Columbus is currently under urban land use whereas 10.3% is covered by a deciduous forest. The region is characterized by a humid continental climate with hot, muggy summers and cold, dry winters.

Predominant upland soils in Columbus are moderately well drained, somewhat poorly drained and very poorly drained soils (SCS 1991). However, pedogenesis and properties of urban soils are disturbed as human activities (i.e., construction activities related to urban development) are also soil-forming factors in urban ecosystems. Thus, some soils in Columbus may contain a significant amount of artifacts (i.e., something in the soil recognizably made or extracted from the earth by humans), or be sealed by technic hard rock (i.e., hard material created by humans, having properties unlike natural rock) (IUSS Working Group WRB 2007). Further, urban soils in Columbus may also include those derived from wastes (e.g., landfills, sludge, cinders, mine spoils and ashes), pavements with their underlying unconsolidated materials, soils with geomembranes and constructed soils from human-made materials.

### 7.5.2 *Study Sites*

The soil C storage was determined in closed canopy forests within two small urban parks in Columbus used for recreational activities. Clinton-Como Park (CP) is

located at the east bank of the Olentangy River north of downtown. In addition to recreational activities, the park is also affected by flooding and sediment deposition, and prior levee construction activities. The CP covers 7.9 ha of which 1.6 ha is covered by closed canopy forest whereas urban lawn with athletic fields, the Olentangy Greenway Bike Trail, a basketball court, a playground, a shelterhouse and single trees cover the other park area. No data are available about the history of this park. At the time of sampling, tree composition included *Acer negundo*, *A. platanoides*, *A. saccharinum*, *Aesculus glabra*, *Ailanthus altissima*, *Asimina triloba*, *Carya cordiformis*, *Fraxinus pennsylvanica*, *Gleditsia triacanthos*, *Juglans nigra*, *Lonicera maackii*, *L. morrowii*, *L. tatarica*, *Morus alba*, *M. rubra*, *Platanus occidentalis*, *Prunus serotina*, *Quercus bicolor*, *Salix nigra*, *Ulmus americana* (pers. comm., Elayna M. Grody, Natural Parks Manager, Columbus Department of Recreation and Parks). Soils are formed on moderately coarse to moderately fine textured recent alluvium, and are classified as Ross silt loam. These soils are prone to occasional flooding (SCS 1991). During soil sampling to 1-m depth, deep roots and fragments of glass and charcoal were observed. River sediment was visible sometimes on the soil surface, and earthworms were abundant. In particular, the southwestern part of CP appeared to be heavily disturbed and compacted by a levee construction.

Soils were also sampled from the Driving Park (DP) located southeast of downtown. The DP covers 9.8 ha, of which 3.1 ha are covered by a closed canopy forest. The remainder areas of the park are covered by urban lawns with athletic fields, tennis courts, a parking lot and the Driving Park Recreation Center built in 1980 (CRPD 2002). The DP received its name from historic past. It was a venue of a large equine racing complex for horses during the nineteenth century, and later for automobiles during early twentieth century (Wikipedia.org). In the early 1990s, DP was one of the chief recreational attractions in Columbus (Columbus Compact Corporation; [www.colcompact.com/pdf/neighprofiles.pdf](http://www.colcompact.com/pdf/neighprofiles.pdf)). It was an amusement center that housed an old grandstand and racing track. The DP featured rides, buggy, bicycle and auto races. A 'driving range' for golfing was also a favorite activity in this area. During the 1920s, single-family homes began to replace the racetrack and summer cottages. By 1922, the DP community was divided into 522 city lots for housing, and the racetrack was completely abandoned during the 1930s. Tree species composition was similar to CP but *P. occidentalis* was absent. Soils are formed in medium-textured and moderately fine textured glacial till, and are classified as Sleeth-Urban land complex. The landscape has a gentle slope of 0–2% in the northern part. The Crosby-Urban land complex has 2–6% slopes in the southern part (SCS 1991). Observations made during soil sampling indicated occurrence of municipal solid waste on the forest soil surface, some construction waste in deeper soil horizons, and fragments of glass and coal. The forest appeared to be disturbed in the eastern part of the DP by a railway dam.

### 7.5.3 Materials and Methods

During summer of 2008, ten randomly distributed, undisturbed, mineral soil samples to 1-m depth were obtained at each urban forest park using a motor-driven soil

column cylinder auger set (Motor breaker Cobra TT, Atlas Copco Construction Tools AB, Stockholm, Sweden; Eijkelkamp Agrisearch Equipment BV, Giesbeek, The Netherlands). After removing the forest floor, soil cores were obtained at distances of at least 1.5 m from the base of trees. The cores (inner diameter 9 cm) were separated into 10-cm increments by trimming both ends with a sharp knife. The entire 10-cm core material was quantitatively transferred into plastic bags, and stored below 8°C pending further processing. Thus, in total 100 mineral soil samples were obtained from each forest patch. The fresh weight of each soil core was recorded. The entire core content was pushed manually through a 2-mm sieve, and living roots and coarse fragments (>2 mm) hand picked, and weighed separately. About 10 g soil, roots and coarse fragments were dried at 105°C for 48 h and the dry weight recorded for determination of moisture content. The remaining bulk soil was air-dried for 2 weeks and then stored at room temperature pending analyses. Air-dried soil samples were ground to pass through a 0.25-mm sieve for chemical analyses.

Soil bulk density ( $\rho_b$ ) was computed as the weight to volume ratio of oven-dried soil corrected for the root and coarse fragment contents (Grossman and Reinsch 2002). Concentrations of C and N in soil samples were measured by the dry combustion method (Vario Max CN Analyzer, Elementar GmbH, Hanau, Germany). The soil C pool ( $\text{Mg ha}^{-1}$ ) for a specific layer of thickness  $d$  (m) was calculated using Eq. 7.1 below (Lorenz and Lal 2007):

$$\text{Mg C ha}^{-1} = \frac{C}{100} \times \rho_b \times d \times \frac{10^4 \text{ m}^2}{\text{ha}} \quad (7.1)$$

where  $\rho_b$  is the bulk density ( $\text{Mg m}^{-3}$ ) of the soil layer corrected for the root and coarse fragment contents, and C concentration is expressed as weight-based percentage (%). All variables were tested for normality (Kolmogorov-Smirnov) and for homogeneity of variances (Levene's test). Because data were normally distributed, no data transformation was required. To compare data among depths within each site and among sites for each depth, differences in means for C and N concentrations and C pools were tested by a one-way analysis of variance (ANOVA;  $P < 0.05$ ). Statistical analyses were done using Statistical Package for the Social Sciences (SPSS for Windows®, Ver. 16.0, Chicago, Illinois).

## 7.6 Results and Discussion

### 7.6.1 Soil Nitrogen and Carbon Concentrations

#### 7.6.1.1 Nitrogen Concentration

The soil N and C concentrations to 1-m depth for CP and DP are shown in Table 7.1. Nitrogen concentrations in both urban forest soils at CP and DP were the highest in 0–10 cm depth (0.39% N and 0.33% N, respectively). Similarly, soil N concentrations under tree cover in urban green spaces in the Front Range of Colorado were

**Table 7.1** Total nitrogen and carbon concentrations to 1-m depth in two urban forest soils in Columbus (N=10; means for each site not sharing a common capital letter are statistically different among depths; means for each depth not sharing a common lowercase letter are statistically different among sites, ANOVA, Student-Newmans-Keuls test,  $P < 0.05$ )

Site	Depth (cm)	Nitrogen (%)			Carbon (%)		
		Min	Max	Mean	Min	Max	Mean
Clinton-Como park	0–10	0.24	0.75	0.39 A,a	3.11	9.71	4.84 A,a
	10–20	0.11	0.41	0.19 B,a	1.52	5.51	2.58 B,a
	20–30	0.08	0.17	0.11 C,a	1.33	2.08	1.60 B,C,a
	30–40	0.08	0.12	0.10 C,a	1.26	1.64	1.41 C,a
	40–50	0.08	0.12	0.10 C,a	1.07	1.70	1.47 C,a
	50–60	0.09	0.14	0.11 C,a	1.10	1.76	1.52 C,a
	60–70	0.09	0.15	0.12 B,C,a	0.99	1.99	1.63 B,C,a
	70–80	0.10	0.15	0.13 B,C,a	1.01	2.16	1.69 B,C,a
	80–90	0.09	0.16	0.13 B,C,a	0.86	2.04	1.63 B,C,a
	90–100	0.08	0.19	0.13 B,a	0.76	2.09	1.60 B,C,a
Driving park	0–10	0.21	0.49	0.33 A,a	2.16	6.06	4.00 A,a
	10–20	0.12	0.33	0.21 B,a	1.08	4.71	2.67 B,a
	20–30	0.09	0.34	0.16 B,C,b	0.65	4.29	1.88 B,C,a
	30–40	0.07	0.31	0.12 B,C,a	0.33	3.42	1.12 C,a
	40–50	0.06	0.23	0.11 B,C,a	0.30	2.41	0.98 C,a
	50–60	0.05	0.31	0.12 B,C,a	0.26	3.24	1.00 C,a
	60–70	0.04	0.27	0.11 B,C,a	0.23	3.06	0.96 C,a
	70–80	0.04	0.18	0.09 C,b	0.25	2.08	0.84 C,a
	80–90	0.05	0.14	0.08 C,b	0.26	2.49	0.97 C,a
	90–100	0.05	0.14	0.07 C,b	0.27	2.52	1.07 C,b

highest in 0–10 cm depth compared to those in 10–20 and 20–30 cm depths (Golubiewski 2006). However, concentrations in 0–10 cm were lower than those in both urban forest soils in Columbus, but comparable to those in unmanaged urban forests in Baltimore, MD (0.16% N; Pouyat et al. 2007). At CP, N concentrations in 10–20 cm depth were higher than those in 20–30, 30–40, 40–50 and 50–60 cm depths, but were comparable to those in 90–100 cm depth. In contrast, soil N concentrations under urban tree cover in Colorado did not differ among 10–20 and 20–30 cm depths (Golubiewski 2006). Differences in N concentrations among other depths at CP were small.

At DP, N concentrations in 10–20 cm depth were higher than those in 70–80, 80–90 and 90–100 cm depths, and differences among other depths were rather small. Concentrations of N in 20–30 cm depth were lower whereas those below 70-cm depth were higher at CP compared to DP. Differences in soil N concentrations among sites for other depths were small. The range in N concentrations until 20-cm depth was wider at CP than at DP, but even larger in 20–80 cm at DP compared to CP. In comparison, soil N concentrations in 10–20 and 20–30 cm depths under urban tree cover in Colorado were lower than those in both urban forest soils from Columbus (Golubiewski 2006).

In summary, profile soil N concentrations were comparable among both urban forests but generally lower at DP below 70-cm depth. Profile soil N concentrations were comparable to those in urban park soils from New York City, NY (Shaw et al. 2009). However, profile N concentrations can be higher in soils used for urban agriculture (Lorenz and Kandeler 2005). Also, soil N concentrations at deeper soil depth can be higher at sites where N-rich material has been buried (Beyer et al. 2001).

### 7.6.1.2 Carbon Concentration

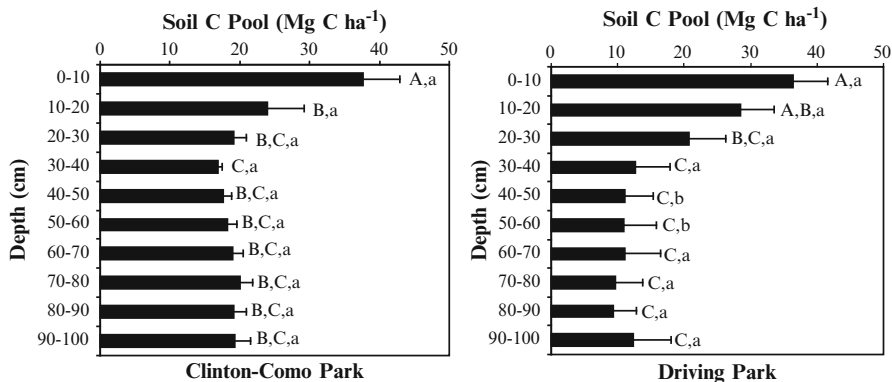
Concentrations of C were the highest in 0–10 cm depth (4.84% C and 4.00% C at CP and DP, respectively) in both parks. Total soil C concentrations were also the highest in 0–10 cm under urban tree cover in Colorado, and higher than in 10–20 and 20–30 cm depths (2.33% vs. 1.38% and 1.35%, respectively; Golubiewski 2006). At DP, C concentrations in 10–20 cm depth were higher than in 30–40, 40–50 and 50–60 cm depths. However, differences among other depths were rather small. In 10–20 cm depth at DP, C concentrations were higher than those below 30-cm depth. Differences among other depths were rather small. Soil C concentrations among sites for each depth were comparable except for C concentrations in 90–100 cm depth which were higher at CP than at DP. The range in C concentrations below 20-cm depth was wider for DP than for CP.

In summary, profile soil C concentrations were comparable among the two urban forest soils in Columbus but higher to 30-cm depth than those under urban tree cover in Colorado. Higher total profile C concentrations have been observed in urban soils containing high amounts of inorganic C derived from soil parent material and/or from buried artifacts (Stahr et al. 2003; Lorenz et al. 2006). Soils containing buried surface layers, buried black carbon (BC) originating from incomplete combustion processes and/or urban soils with C-containing artifacts such as ash, asphalt, coal and slag have also high soil profile C concentrations (Shaw et al. 2009).

## 7.7 Soil Carbon Storage

The depth distribution of the soil C pool to 1-m depth for CP and DP is shown in Fig. 7.1. At CP, the C pool was the highest in 0–10 cm depth (37.6 Mg C ha<sup>-1</sup>), and higher in 10–20 cm compared to that in 30–40 cm depth (24.0 Mg C ha<sup>-1</sup> vs. 16.8 Mg C ha<sup>-1</sup>, respectively). Differences in soil C pool were rather small among other depths. In total, 210.9 Mg C ha<sup>-1</sup> were stored to 1-m depth at CP.

At DP, the C pool in 0–10 and 10–20 cm depths (36.4 and 28.4 Mg C ha<sup>-1</sup>, respectively) was higher than below 30-cm depths. Differences in soil C pools among other depths were rather small. In total 162.6 Mg C ha<sup>-1</sup> were stored to 1-m depth at DP, but this pool was similar to that stored at CP (ANOVA, Student-Newmans-Keuls test,  $P < 0.05$ ). Also, in 40–50 and 50–60 cm depths, soil C pools at CP were



**Fig. 7.1** Soil carbon pool ( $\text{Mg C ha}^{-1}$ ) to 1-m depth at two urban forests in Columbus ( $N=10$ ;  $\pm\text{SD}$ ); means for each site not sharing a common capital letter are statistically different among depths; means for each depth not sharing a common lowercase letter are statistically different among sites, ANOVA, Student-Newmans-Keuls test,  $P<0.05$ )

higher than those at DP. Differences among sites for other depths were small. The variance in soil C pools until 20-cm depth was comparable among sites, and higher at DP below this depth.

In summary, soil C pools to 1-m depth did not differ among both urban forest soils from Columbus, but were higher than those in urban soils from wooded areas in New York City (Shaw et al. 2009). In addition to high concentrations of inorganic C, BC and C-containing artifacts, and differences in bulk density contributed to high profile C pools in urban soils of the Columbus city.

## 7.8 Conclusions

During the past 200 years the original forest cover of Ohio had been converted to other land uses, particularly agricultural land use. In recent decades, however, conversion of farmland to urban land uses is rapidly increasing. Such a rapid urbanization is accompanied by planting of urban trees, and the establishment of urban forests but their soil C storage has not been previously assessed. Among both urban forests in Columbus, the soil C pool to 1-m depth did not differ, but was higher than in soils of wooded areas in New York City. However, additional analyses are required to obtain soil organic carbon (SOC) concentrations for the forests in Columbus by correcting for inorganic C and geogenic C or coal impurities. Such studies must also assess the C storage of the entire soil profile to about 2-m depth. Specifically, tree roots and their associated microorganisms are the major sources for profile SOC, and tree roots reach 4.6-m depth on average (Canadell et al. 1996).

Other profile SOC sources are dissolved organic carbon (DOC) and bioturbation (Lorenz and Lal 2005). In comparison, non-urban forests soils in Ohio store 49–239 Mg SOC ha<sup>-1</sup> to 1-m depth, and up to ~1,150 Mg SOC ha<sup>-1</sup> may be stored in Histosols covered by forest (Tan et al. 2004). Further, urban forests in Atlanta store 77 Mg SOC ha<sup>-1</sup> and those in Baltimore 115.6 Mg SOC ha<sup>-1</sup> to 1-m depth (Pouyat et al. 2006, 2009).

## 7.9 Outlook

Emissions from the combustion of fossil fuels and land-use changes increase the atmospheric CO<sub>2</sub> levels causing ACC. For mitigating ACC, the terrestrial C storage in vegetation, litter and soil in urban areas must be increased by the process of C sequestration. Previous studies on Ohio's urban environment have assessed key environmental indicators such as conversion of open spaces, changes in farmland area, loss of wildlife habitat, number of endangered plant species, brownfields, air pollutants, and lake quality etc. (Kaplan et al. 2001). However, terrestrial C sequestration studies in urbanizing landscapes of Ohio focused primarily on the effects of urban-residential developments in suburban and exurban areas outside of incorporated urban areas (Wang and Medley 2004). In particular, the incorporation of woodlots with deep soil profiles is recommended for C conservation and storage across regional landscapes in the Midwest. Urban forests of Ohio have been less well studied. For example, tree-related benefits such as the amount of stormwater runoff and its rate of discharge can be greatly increased by broadened urban forest management in Dayton (Sanders and Stevens 1984; Sanders 1986). Another area of research in urban forests in Ohio is the damage to ash trees (*Fraxinus* spp.) caused by infestation with the emerald ash borer (*Agrilus planipennis* Fairmaire, 1888) as ash trees are one of the more widely planted trees in urban areas (MacFarlane and Meyer 2005). Also, birds in urban forests in Columbus and their migratory behavior and conservation has received some attention (Matthews and Rodewald 2010; Sundell-Turner and Rodewald 2008). However, previous studies have indicated that urban forests in Ohio sequester atmospheric CO<sub>2</sub>. The present study also indicates that urban forest soils in Ohio store appreciable amounts of C. Thus, C management may be an important ecosystem service for Ohio's urban forests (Williams 2010). However, the net effects of urban forests on the local and regional climate, as influenced by C sequestration and dynamics, must be assessed by studying: (i) cooling through reduction of atmospheric CO<sub>2</sub> concentration directly by C sequestration and indirectly by replacing fossil fuels with urban woody biomass (de Richter et al. 2009), (ii) cooling or heating through emissions of biogenic volatile organic compounds (BVOCs) and their effects on aerosol, ozone and cloud formation (Goldstein et al. 2009) and, (iii) biophysical effects (albedo, hydrology) of urban forest cover.

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