Rattan Lal · Bruce Augustin *Editors*

Carbon Sequestration in Urban Ecosystems

Springer

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Foreword

Urbanization, a principal land use during the twenty-first century, is an anthropogenically driven and a rapid transformation of ecosystems. While more than 50% of the world's population already lives in urban centers, urban encroachment of prime farmlands is caused by a rapid population growth. By 2050, as much as 69% of the world population may live in urban centers. It is estimated that providing accommodation and the supporting infrastructure to one million people requires 40,000 ha of land. Thus, the estimated global population growth rate of 70–80 million people per year needs an additional land area of three million hectares (Mha). Total land area in urban centers in the U.S. is estimated at 24 Mha or 2.62% of the total land area. Such a rapid urbanization has also increased demand for natural resources, with a large ecologic foot print. Further, continued loss of prime farmland may exacerbate food prices, create agricultural shortages across the world, and lead to food price inflation. Urban encroachment has numerous social, economic, political and ecologic implications, including reduction in ability of agroecosystems to produce adequate amount of food and fiber for the ever increasing world population.

Megacities and growing urban centers also use numerous resources including energy, minerals, transport fuel, water and food, and generate a large amount of waste. This drastic ecologic transformation and consumption of natural resources is a principal anthropogenic driver of global change including the global warming. Of the total earth's ice-free land area of 11.3 billion ha, about 3% (338 Mha) is under urban land use. However, these areas are major sources of emission of greenhouse gases. Yet, judicious management and restoration of urban ecosystems can off-set some anthropogenic emissions and also generate essential ecosystem services.

Urban centers consist of build up areas (under buildings, concrete and asphalt) and also green areas (under lawns, shrubs, trees, forests and agriculture or horticultural gardens). While efficient use of energy, water, and minerals in build up areas is extremely important, sustainable management of green areas is essential to restoring the ecosystems C budget. Improved management of green areas can sequester C in the above and below-ground biomass, increase soil organic C pool and improve its depth distribution in favor of translocation of C into the sub-soil layers, reduce emissions of N_2O and CH_4, and off-set some anthropogenic emissions.

It is argued that the twenty-first century is the Century of the Cities, because of their major impact on global process and significant issues of the twenty-first century. Therefore, a workshop was organized at the campus of The Ohio State University (OSU) entitled, "Carbon Sequestration in Urban Ecosystems" on 14th April 2010. Jointly organized and sponsored by OSU and Scotts Co., the workshop was attended by about 50 participants from around the country. This volume is based on papers presented at the workshop. In addition, some other renowned researchers/authors were invited to contribute additional chapters to address regions or themes not covered by the workshop participants. Principal objectives of the workshop, and of this volume, are to:

- Assess the effects of urbanization on national and global C pools and fluxes,
- Determine credible estimates of C pool and fluxes in turf grass, home lawns, urban agriculture and gardens, urban forests and trees and other land use,
- Evaluate the role of turf grass and lawns on emissions of greenhouse gases (CO_2, CH_4, N_2O),
- Study coupled cycling of C with that of H_2O, N, P, etc. in relation to soil quality in lawns and turf grass systems,
- Identify best management practices for diverse land uses within urban green spaces,
- Develop strategies which promote adoption of best management practices to enhance sequestration of C within green areas of urban ecosystems, and
- Identify management systems which reduce emissions of greenhouse gases from urban land systems, and enhance use efficiency of inputs.

These objectives have been summarized into some basic questions which need to be addressed. Some important questions are:

- How much and by what processes does the urbanization influence the global carbon cycle?
- What is the quantitative estimate of diverse green areas (i.e., lawns, turfs, forests, agriculture) in urban centers, and how can the green areas be sustainably managed?
- How does management influence C budget of the "green city areas" within urban ecosystems?

The 19-chapter volume specifically addresses the objectives and question listed above. Specific topics addressed include: (i) adapting urban land use to climate change, (ii) managing urban forests and lawns to sequester carbon, (iii) assessing C pools, and gaseous fluxes in urban ecosystems, and (iv) promoting urban agriculture.

The editors thank all the authors for their outstanding contributions to this volume. Thanks are also due to the staff of Springer Verlag for their timely efforts in publishing this information, and making it available to scientists, practitioners, and general public. Editors specifically acknowledge support received from the staff and students of the Carbon Management and Sequestration Center of The Ohio State University, and the staff of the Scotts Co. for their help and support. We especially thank Ms. Theresa L. Colson for her efforts in handling the flow of the chapters

included here. Help received from Ms. Allie Curry in checking the references and formatting is greatly appreciated. The efforts of many others were extremely important in publishing this relevant and important volume.

Columbus, OH

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Chapter 1
Urban Ecosystems and Climate Change

Rattan Lal

Abstract More than 50% of the world population (~3.5 billion) lives in urban areas, and the relative magnitude will increase to 60% by 2030. The highest rate of urbanization in the world is in Latin America, and in the emerging economies of China and India. Urban ecosystems, covering ~3% of the terrestrial land area, strongly influence biogeochemical cycles of elements (C, N) and water, and alter regional and global climate through gaseous emissions. Yet, urban lands are an important C pool with a C density as high as 20–40 kg C m^{-2}. Because of intensive management, the above ground net primary productivity can be 300–400 g C m^{-2} $year^{-1}$. Principal components of the ecosystem C pool include urban forests, lawns and turfs and recreational grounds, and soil C pool. The net ecosystem C pool can be enhanced by reducing the hidden C costs associated with management. Urban agriculture is gaining importance and adds to multifunctionality of urban landscapes. Urban ecosystems have a large technical potential to sequester C in soils and biota through judicious management. While sources of gaseous emissions must be reduced, C sink capacity of urban lands can be enhanced through adoption of recommended management practices.

Keywords Urban agriculture • Soil carbon in urban land • Green roofs • Urban soil quality • Urban forests

List of Abbreviations

ACC	abrupt climate change
ANPP	above-ground net primary productivity
C	carbon

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GCC global carbon cycle
GHGs greenhouse gases
HCC hidden carbon costs
NEE net ecosystem exchange
NPP net primary productivity
SOC soil organic carbon
USDA United States Department of Agriculture

1.1 Introduction and Rationale

The world population increased from 2.53 billion in 1950 to 6.91 billion in 2010, an increase of 173% (UN 2008). Between 2010 and 2050, the world population will increase by another 32% to 9.15 billion. In comparison, world's urban population increased from 165 to 215 million in 1900 to 729 million in 1950 and 3.486 billion in 2010, 21 times compared with 1900 and 3.8 times in comparison with 1950. Between 2010 and 2050, world's urban population will increase by another 80% to 6.29 billion. By 2050, 68.7% of the world population will live in urban centers (Fig. 1.1a, b; UNPD 2011). These trends indicate a drastic shift to urban living since 1900 (Grimm et al. 2008; Seto and Satterthwaite 2010). Urban population, as a percentage of the world total population, was 10–13% in 1900, 28.8% in 1950, 50% in 2009, and is projected to be 60% in 2030 and 68.7% in 2050 (Fig. 1.1c; UNPD 2011). The rate of urbanization is very rapid in less developed and developing countries (Seto and Shepherd 2009; Table 1.1). Similar trends in urbanization are observed in USA and North America. The urban population had increased from 130 million in 1960 to 259 million in 2010 in the U.S.A., and 143 million to 281 million in North America (Table 1.2). The urban population in the state of Ohio increased from 5.6 million (70.2%) in 1950 to 8.9 million (77.4%) in 2009 (Table 1.3).

Concentration of world population in large urban centers or mega cities (i.e., Mexico, Rio, Mumbai, Kolkata) strongly impacts regional and global climate (Tollefson 2010a, b) because of drastic perturbations of land use and the attendant changes in biogeochemical cycles of H_2O and elements (C, N, P, S), energy use, radiation budget, and the heat island effect (Buckley 2010). Urban population growth and export of agricultural products are also major factors causing tropical deforestation (DeFries et al. 2010). Thus, the objective of this chapter is to outline the impact of urbanization on ecosystem functions, but specifically on the global C cycle in relation to climate change adaptation and mitigation (Table 1.4).

1.2 Urban Ecosystems and Climate Change

Urban lands constitute an important component of the global land use and area change matrix (Holmgren 2006), thus, must be considered in all climate and environmental issues. Global urban centers cover ~3% of the terrestrial land area

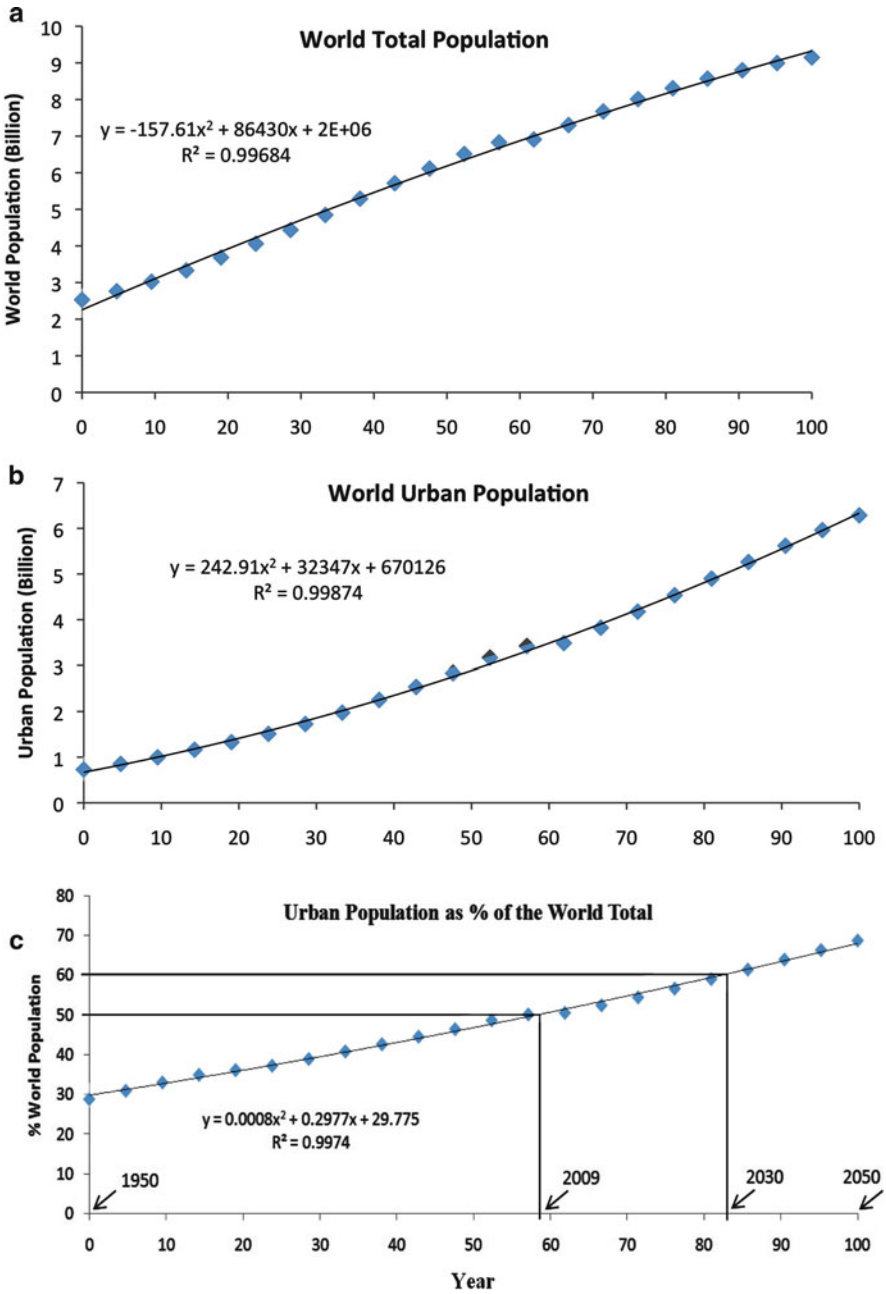

Fig. 1.1 (**a**) trends in world population (**b**) world urban population (**c**) % urban population from 1950 to 2050 (Redrawn from UNPD 2011)

Table 1.1 Continental distribution of urban population (UN 2008)

Region	Urban population (10^9) 2007	2025	2050
World	3.29	4.58	6.40
Africa	0.37	0.66	1.23
Developed regions	0.91	0.99	1.07
Less developed regions	2.38	3.59	5.33
China	0.56	0.83	1.01

Table 1.2 Urban population in North America (10^9) (UN 2008)

Year	USA	North America
1960	130	143
1970	155	171
1980	170	189
1990	193	214
2000	225	150
2010	259	286
2020	291	321
2030	318	351
2040	343	378
2050	364	401

Table 1.3 Urban population in Ohio (Ohio Data Center 2010)

	Population (10^9)		
Year	Total	Urban	Urban (% of total)
1950	7.9	5.6	70.2
1960	9.7	7.1	73.4
1970	10.7	8.0	75.3
1980	10.8	7.9	73.3
1990	10.8	8.0	74.1
2000	11.4	8.8	77.4
2009	11.5	8.9	77.4

Table 1.4 Energy use in urban households (Williams 2008)

			Residential energy use (10^6 BTU)			
Unit	Parameter	Area (ft^2)	Heating	Cooling	Other	Total
Single-family	Detached	2,368	45.7	8.0	44.0	89.7
Single-family	Attached	1,121	36.6	6.1	36.7	79.4
Apartment	Multifamily	1,067	24.5	6.4	28.3	59.2

(Grimm et al. 2008). However, urban ecosystems are the major source of global emissions including 78% of total C emission. Urban centers also use 60% of the residential water consumption and 76% of industrial wood consumption (Grimm et al. 2008). Consequently, there is a drastic transformation in resource use because of the strong impacts of urbanization on the environment (Fig. 1.2). Principal among these impacts are physical alteration because of (i) transformation of landscape,

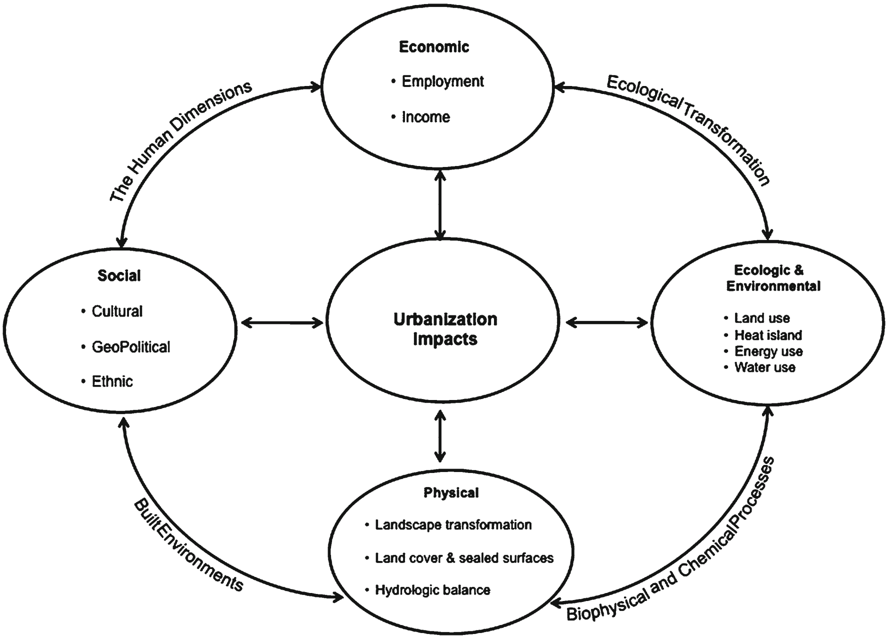

Fig. 1.2 Ecologic, economical, physical and social impacts of urbanization in the context of climate change

(ii) changes in land cover and sealed surfaces, and (iii) alternations in the hydrologic balance. Urbanization in the desert or dry regions is a principal challenge of water sustainability because urban cities are vulnerable to water scarcity (Gober 2010). There is a distinct water–energy relationship in desert urban centers. These physical changes lead to ecologic and environmental impacts including those on land use, heat island, and energy and water use. These impacts are also related to the human dimensions or social issues as influenced by job opportunities and employment, and thus to changes in income or economic affluence of the urban population. Another important aspect of the human dimensions of the urbanization involves changes in social and community structure encompassing those related to cultural, geopolitical and ethnic mix in urban population. These impacts, outlined in Fig. 1.2, strongly alter the biogeochemical cycles, especially the C cycle, with an attendant changes in both regional and global climate.

1.3 Urban Land Use and Climate Change

In comparison with natural and agroecosystems or forest plantations, urban ecosystems have a distinct biogeochemistry (Kaye et al. 2006). Principal biogeochemical controls include impervious surfaces, engineered aqueous flow paths, drastic

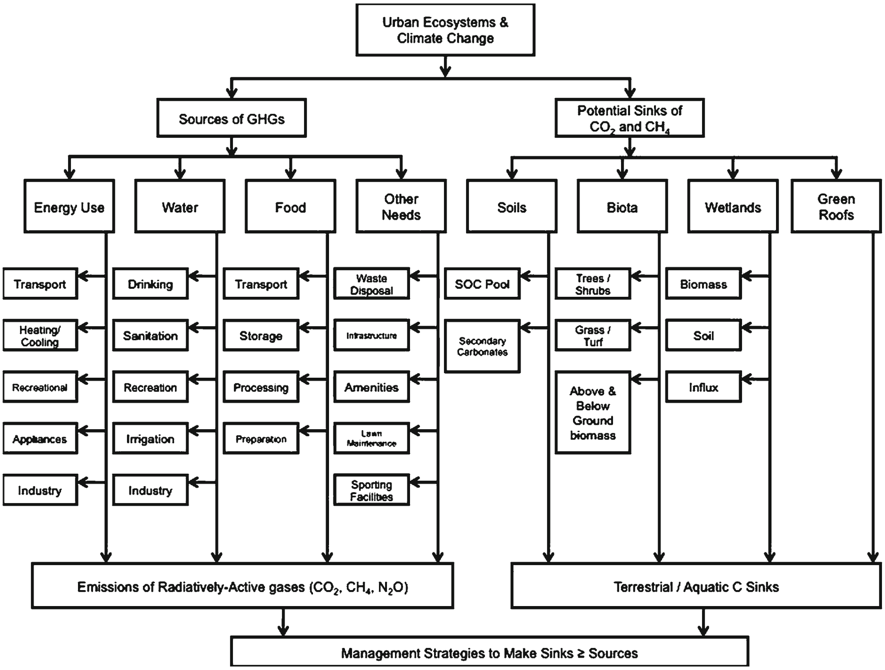

Fig. 1.3 Sources and potential sinks of greenhouse gases in urban ecosystems (*SOC* soil organic carbon)

landscape alterations, and perturbed ecological processes (Jenerette et al. 2006). Thus, urbanization influences the global C cycle (GCC) through alterations in sources and sinks of C (Fig. 1.3). Urbanization changes the characteristics of the vegetation and soils, both of which are a distinct component of the GCC (Svirejeva-Hopkins et al. 2004). Through alteration of C pools and their dynamics (flows), it influences the climate at local, regional and global scales. Urbanization is a process of ecological transformation (Huang et al. 2010).

The principal sources of CO_2 and other greenhouse gases (GHGs) in urban ecosystems are attributed to the concentration of transport and industry in these sites. The poor air quality in megacities (Parrish and Zhu 2009) is attributed to emission of a wide range of trace gases from industrial sources. Important among these traces gases are NO, NO_2, O_3. SO_2, HNO_3 and a range of organic acids (Grimm et al. 2008). There are also elevated concentrations of CO_2, CH_4 and O_3 compared with rural areas. Air pollution also affects the net primary productivity (NPP) and indirectly influences the GCC. Other principal sources of GHGs in urban ecosystems are from infra-structure, sporting facilities, lawn maintenance, and other amenities.

It is estimated that by 2015, there will be 236 cities in the world with a population of 10 million or more. A city of this size requires about 6,000 tons of food per day. Thus, a large number of plant nutrients are transferred into the cities, which

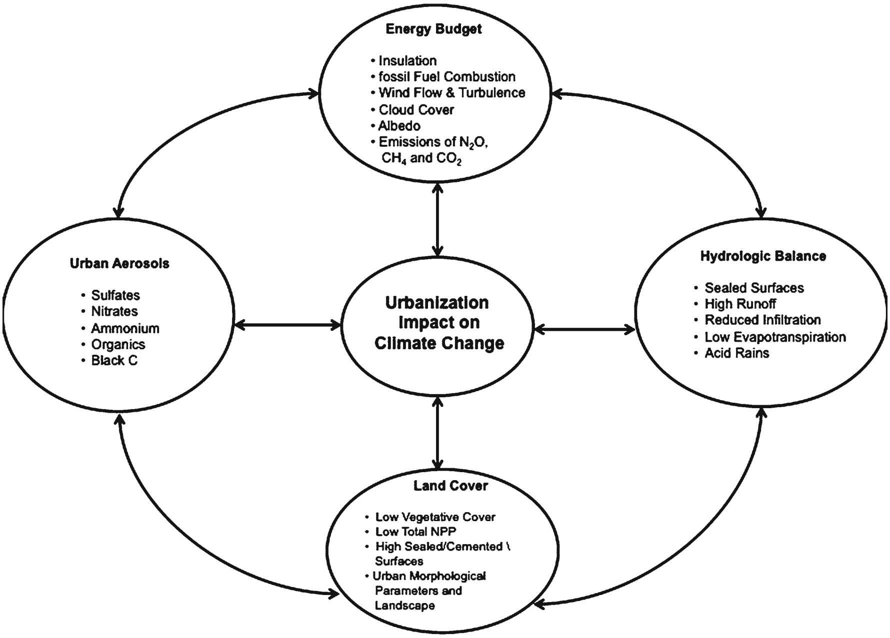

Fig. 1.4 Urbanization effects on climate change through alterations in energy budget, hydrologic balance, land cover and aerosols

accumulate N, P, metals and other pollutants. Megacities are considered similar to an organism that consumes a large amount of food and generates a considerable quantity of waste (Walsh 2002), and the latter is a principal cause of environmental pollution and of the abrupt climate change (ACC). Waste disposal is an important factor, and landfills are a major source of GHGs.

There are also numerous potential sinks of GHGs within the urban ecosystems (Fig. 1.3). Important among these are soils, biota, wetlands and green roofs. With judicious management of soils and trees (along with wetlands and green roofs etc.), urban ecosystems can be a major sink of CO_2 and other atmospheric constituents.

The overall effects of urbanization on climate change, outlined in Fig. 1.4, include those due to drastic alterations in the energy budget, hydrologic balance, land cover and NPP, and the aerosols. The effects of aerosols on climate are complex (Seto and Shepherd 2009) and can be positive or negative, and direct or indirect. Some aerosols have a cooling effect (i.e., sulfates). Most urbanization occurs on highly fertile lands/soils. Thus, there is a strong negative effect of urban encroachment on NPP and the ecosystem C budget. High emissions of N_2O (and CH_4 from waste and sewage) create high radiative forcing in urban environments. The reduction in NPP through urbanization is exacerbated by increase in deforestation (DeFries et al. 2010). Indeed, there are numerous interactions between urbanization and climate change and other environmental factors (Seto and Satterthwaite 2010).

Creation of sealed surfaces has buried a considerable amount of soil C beneath the cemented or covered surfaces. Yet the terrestrial C stored in human settlements has not been adequately quantified. Churkina et al. (2009) estimated that the amount of terrestrial C stored in urban settlements can be as much as 23–42 kg C m^{-2} compared with 7–16 kg C m^{-2} in rural areas and 4–25 kg C m^{-2} in tropical rainforests. In 2000, the terrestrial C pool in urban lands in the contiguous USA was 18 Pg or 10% of the total C pool (Churkina et al. 2009). Of this, 11.5 Pg (64%) was stored in the urban soils. Thus, management of soil C pool in urban ecosystems is important to the GCC and ecosystem C budget (Pavao-Zuckerman 2008). Thus, protecting and enhancing the terrestrial C pool in urban ecosystems is an important strategy to mitigating its average impacts on the environment, and for sustainable development (Rodriguez and Bonilla 2007).

1.4 Sources of Gaseous Emissions

Two principal aspects of urbanization, as major sources of gaseous emissions, are: consumption of natural resources (i.e., fossil fuel, forest products, water), and transformation of land and ecosystems (i.e., deforestation, drainage of wet lands, landscaping, etc.). It is the large demand for energy and natural resources which exacerbate emissions of CO_2, CH_4, N_2O and other trace gases into the atmosphere. Urban sprawl as a result of human population growth is an important source of increased atmospheric CO_2 (Shao et al. 2008).

Urban centers, and especially the megacities, are major sources of GHGs. A wide range of emission sources in urban ecosystems include use of electricity and natural gas for residential, commercial and industrial activities. City growth affects the environment directly and indirectly (Bettencourt and West 2010), and has been a source of much pollution. In addition, there is a large consumption of petro-fuels (diesel, gasoline, jet fuel, etc.) for vehicles (cars, trucks) used in transport (Hillman and Ramaswami 2010). Trucking is a dominant freight mode in North America and elsewhere. In addition, a vast amount of cement used in urban centers is also a major source of CO_2.

1.5 Urban Land Use and Carbon Sinks

Management of urban landscape is important to moderating the GCC. Urbanization affects the sustainable use of natural resources (Lyons 1997). The twenty first century is considered as The Century of the City, and cities are leading the way in climate-change action plan (Armstrong and Spiller 2010; Rosenzweig et al. 2010; van Noorden 2010c), and in institutions (Butler 2010). The United States Department of Agriculture (USDA) also has a program in urban ecosystem management. With a large proportion of the terrestrial C pool stored in urban settlements (both pedologic and biotic), a sustainable management of this pool is essential to moderating the GCC, and mitigating ACC through enhancement and sequestration of C in urban landscapes. The strategy of enhancing the C pool in urban ecosystems is especially relevant because anthropogenic drivers dominate natural drivers in the control of the terrestrial C pool in urban ecosystems (Pouyat et al. 2009). The intense management (anthropogenic drivers), strongly impacts the above-ground net primary productivity (ANPP; 383 ± 11 g C m^{-2} year^{-1}) which can be 4–5 times greater than those of wheat (*Triticum aestivum)* or shortgrass steppe but lesser than that of corn (*Zea mays)* (537 ± 44 g C m^{-2} year^{-1}) (Kaye et al. 2005). Furthermore, soil respiration ($2{,}777 \pm 273$ g C m^{-2} year^{-1}) and total below-ground C allocation ($2{,}602 \pm 269$ g C m^{-2} year^{-2}) in urban ecosystems, can be 2.5–5 times more than any other land use (Kaye et al. 2005).

The large terrestrial C pool in urban ecosystems comprises of four components: (i) urban forests, (ii) grasslands, including turfs, lawns and recreational grounds, (iii) soil C pool, and (iv) urban agriculture. Urban forests, constituting an important C pool in these ecosystems (McPherson et al. 1997), are characterized by a unique ecosystem structure and function in relation to the rural forest stands: heat island effect, pollution, heavy metals, etc. (McDonnell et al. 1997). Pouyat et al. (2002) reported that urban forest stands have significantly higher SOC densities (kg m^{-2} to 1-m depth) than the suburban and rural stands. Similarly, urban grasslands (i.e., lawns, turfs) are important to the GCC. There are approximately $163{,}800 \pm 35{,}850$ km^2 of the land area planted to turf grasses in the continental U.S. (Milesi et al. 2005). These well-fertilized and adequately irrigated ecosystems are three times larger than the irrigated cropland area in the U.S. Thus, judicious management of turfs and lawns (grasslands) is also an important strategy of moderating the terrestrial C pool in urban ecosystems, and to influence the GCC. Urban green spaces store a large amount of C pool, even more so than agricultural or native grasslands on unit area basis, or as C density (Golubiewski 2006). At equilibrium, within 25–50 years after establishment of lawns and turfs, especially in degraded agricultural soils, the SOC pool in turfs can exceed that in natural grasslands (Golubiewski 2006).

Several studies have shown that well-watered and adequately fertilized turf grasses are a net sink of atmospheric CO_2. However, hidden C costs (HCCs) of management and emission of N_2O and CH_4 must be carefully addressed to optimize the net C sink of urban lawns and turfs. Excessive HCCs of management (i.e., fertilizers, pesticides, irrigation, mowing) can negate the gains in ecosystem C pool. The potential net ecosystem exchange (NEE) of total land area in the U.S. is estimated at 17 Tg C year^{-1}. It would thus require up to 695–900 l of water per person per day depending on the soil, climate, mode of irrigation, grass species, etc. (Milesi et al. 2005). Irrigation and use of fertilizers also cause nitrate leaching and N_2O emissions from urban grasslands (Groffman et al. 2009), which must be carefully managed.

1.6 Enhancing Soil Carbon Pool in Urban Ecosystems

Land-based C sinks in urban ecosystems include those in biomass and soils. High SOC densities (15.5 ± 1.2 kg m^{-2}) observed in residential areas (Pouyat et al. 2002) are attributed to several factors including: (i) large productivity related to high inputs,

(ii) lack of soil disturbance, and (iii) return of biomass (clippings). Thus, the SOC pool in urban lawns can be accentuated by management of these factors to create a positive soil C budget through sequestration. In most turf grasses (i.e., golf courses, home lawns), C sequestration in soil can continue up to 30–45 years after establishment, although most rapid increase occurs during the first 25–30 years (Qian and Follett 2002). Further, low density residential and institutional land uses may have 38% and 44% higher SOC densities than the commercial land use types, respectively. The SOC density in urban ecosystems also depends on the degree of physical perturbation, landscaping, nature of the soil fill, and management including water and N (Pouyat et al. 2002).

The large input of reactive forms of N, water, and other inputs are important to the rate, magnitude, and duration of SOC sequestration in urban lawns. In an arid urban ecosystem of Arizona, Zhu et al. (2006) observed that its average soil N concentration was 3.23 g m^{-2}, indicating net immobilization of inorganic N in the desert environment, and posing a threat to surface and groundwater contamination and to emission of N_2O in the atmosphere. In addition to the effect of climate, the relative magnitude of immobilization and mineralization of N also depends on the addition of biomass-C from the clippings. Addition of clippings can increase the potential nitrification in young turf grass, and the response of soil microbial biomass and N mineralization and immobilization to clipping addition is independent of indigenous soil and microbial attributes (Shi et al. 2006).

Climate, temperature and precipitation, also affected the SOC pool in urban lawns. The SOC density in urban lawns, ranging from 0.3 to 12.7 kg m^{-2}, varies strongly across climatic gradients. In general, cities located in warmer and/or drier climates may have somewhat higher SOC pools in the past than in pre-urban development stage (Pouyat et al. 2006). The ratio of above-ground to below-ground estimates of C pool may be as high as 2.8–3. The SOC pool in urban soils also depends on several unique features such as creation of novel soil types, drastic alterations in soil properties brought about by the historic land use, and domination of non-native plant species (Pavao-Zuckerman 2008). It is the deliberate and intense manipulations of urban soils that create a large gross C sink capacity, although the net sequestration may be much lower because of the HCCs of inputs. For example, Pouyat et al. (2009) observed that within Baltimore, turf grass had almost 2-fold higher SOC density at 0–0.2 and 0–1 m depth than in rural forests. Similar observations were made for the SOC pool in turf grass in Denver compared with that in the short grass steppe soils.

While climate (temperature and precipitation) is important, its effect on SOC pool is masked by intense management. Because anthropogenic drivers dominate the ecosystem processes, turf grass systems have similar but high SOC densities across climates, parent material and topography (Pouyat et al. 2009). Such high SOC densities in urban turfs despite diverse environments are due to greater management efforts or anthropogenic manipulations. Nonetheless, the C footprint of urban management is high because of intense management. Therefore, management (i.e., mowing, fertilizer, irrigation, species) is important to enhancing the net rate and magnitude of SOC pool in urban ecosystems.

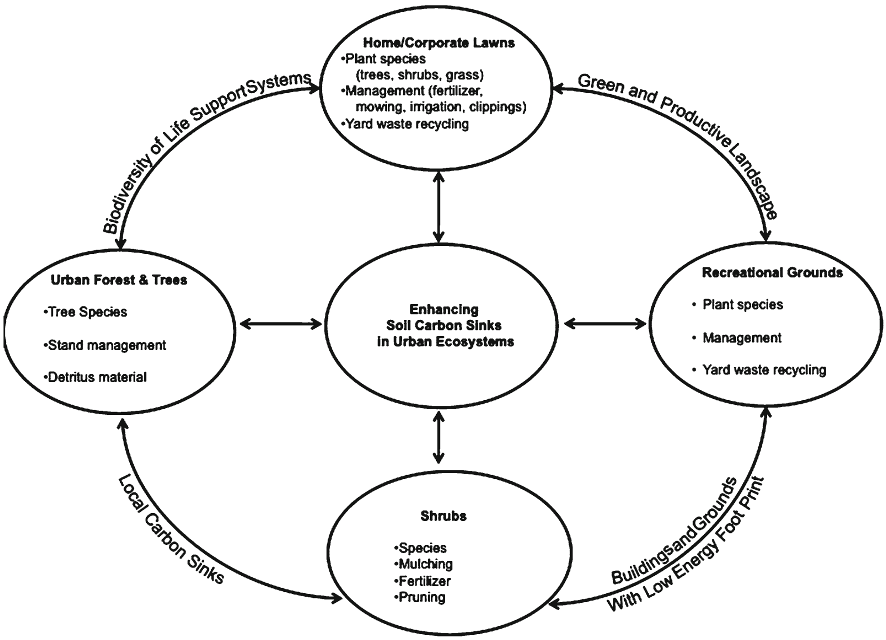

Fig. 1.5 Strategies to enhance soil-based sinks in urban ecosystems, the soil C pool is an important component of home/corporate lawns, recreational/sport grounds, forests and trees, and ornamental shrubs

There are numerous strategies to enhance the soil-based sinks in urban ecosystems (Fig. 1.5). The general principle is to enhance NPP per unit input of fertilizers, pesticides, water and energy so that losses are minimized and the use efficiency is accentuated. Management strategies include those which create a positive net C budget. These strategies include: a choice of appropriate plant species, fertilizer rate and timing of application, mowing frequency and management of clippings, and household/yard waste recycling and management for home/corporate lawns. The strategies for recreational grounds, urban forests, and shrubs also include choice of plant species, management and waste recycling.

Design and construction of green buildings is an important aspect of improving the urban environments. The goal is to make most corporate buildings as C-neutral and most residential buildings as energy efficient. Creating green roofs, an ancient strategy adopted by Vikings (Fig. 1.6) and even now in Europe (Fig. 1.7), is important to enhancing the net ecosystem C pool of urban landscapes. All components of urban landscape must be sustainably managed. Nourishing urban environment is a new paradigm (Knight and Riggs 2010).

Fig. 1.6 Green roofs was the tradition of Vikings since the settlement of Iceland

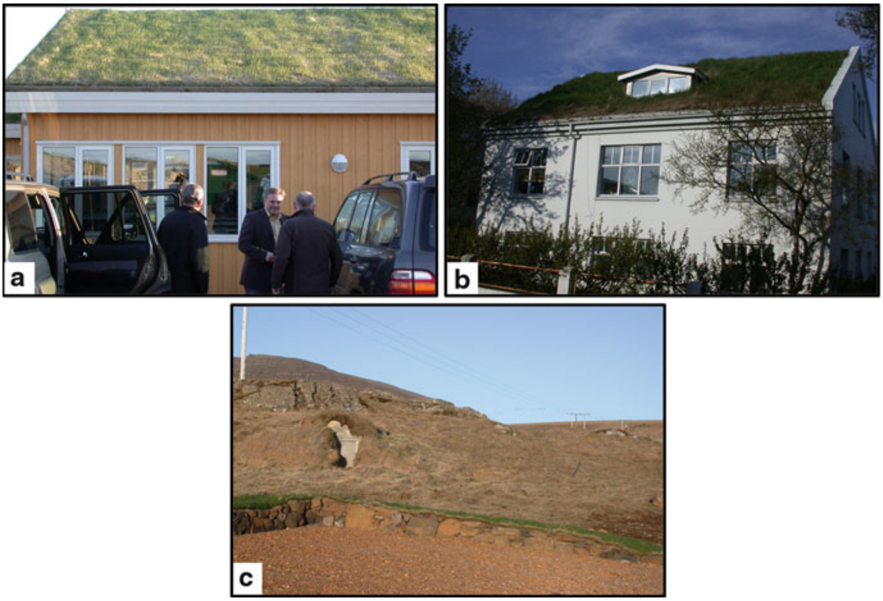

Fig. 1.7 Green roofs are also widely used in modern European architecture: (**a**) a commercial building (**b**) residential building and (**c**) farm storage facility

1.7 Urban Agriculture

Urban ecosystems are characterized by multi-functionality (Pearson 2010; van Leeuwen et al. 2010). Important multifunctions of urban ecosystems comprise their positive impacts on ecologic, economic, social, land use planning and other multi-dimensional aspects (Table 1.5). Urban green-spaces are important to enhancing the aesthetic environments, promoting recreation and leisure, creating jobs and tourism, and producing food for the urbanites (Fig. 1.8). Urban green spaces for these multi-functions have been used throughout the human history over millennia (van Leeuwen et al. 2010).

Similar to conventional agriculture, urban agriculture also has three distinct but related components: social, economic and environmental (Pearson et al. 2010).

Table 1.5 Multi-functionality of urban land use

Values		Description
1.	Ecologic	Intrinsic natural value; genetic diversity value; life-support value
2.	Economic	Market (crop production, carbon sequestration)
3.	Social	Recreational, aesthetic, cultural
4.	Planning	Instrumental, synergetic, competitive
5.	Multidimensional	Scientific, policy

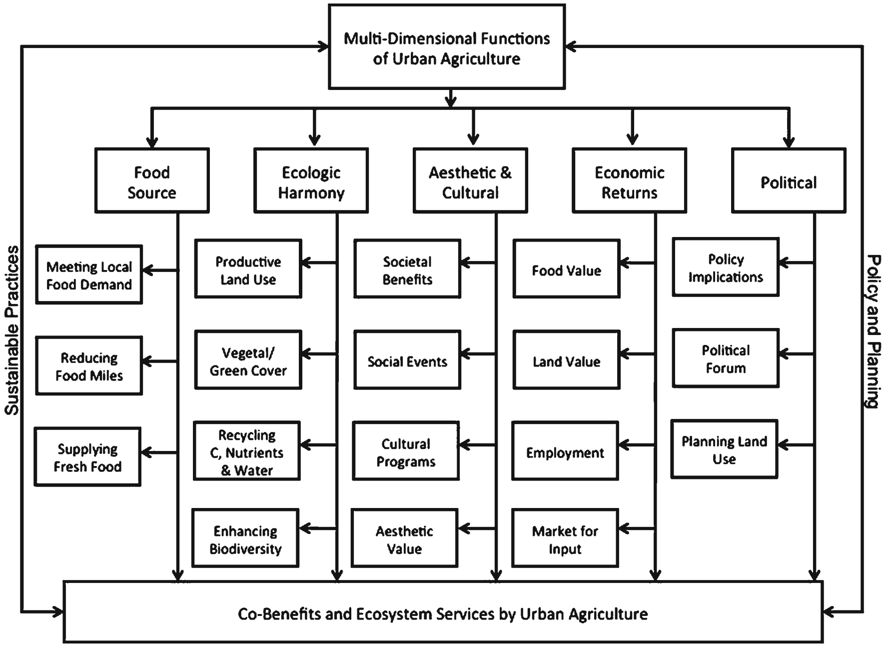

Fig. 1.8 Role of urban agriculture in sustainable management of urban ecosystems

The social component consists of food security, gender equity, and nutritional/diet quality, etc. The economic component entails employment, a productive land use, reducing food-miles, etc. Important environmental issues comprise waste recycling, C sequestration, and moderation of micro-climate.

In addition, and also in accord with the multidimensional aspects, urban agriculture substantially contributes to land use planning, synergism and competitiveness, scientific data, and policy relevance (van Leeuwen et al. 2010). Furthermore, urban agriculture comprises specific features which are distinct from the large-scale, commercial and intensive agriculture (Beus and Dunlap 2010; Sumner et al. 2010). Six among principal comparative features of these two agricultural models are: (i) centralization vs. decentralization, (ii) dependence vs. independence, (iii) competition vs. community, (iv) with or against nature, (v) monoculture vs. diversity, and (vi) exploitation vs. restoration (Sumner et al. 2010). The focus is on local food production and direct marketing (Mason and Known 2010). The direct marketing involves open or a roadside stand, pick your own, farmers' market, etc. The goal is to increase the demand for local and regional food, improve financial viability of the household, and diversify local livelihoods (Mersen et al. 2010). Condon et al. (2010) listed ten merits of urban agriculture: (i) economic, (ii) low cost, (iii) job opportunities, (iv) societal benefits, (v) social integration, (vi) environmental compatibility, (vii) food quality as healthy and fresh foods, (viii) favorable urban environment and living, (ix) diverse and multi-dimensionality, and (x) enhancement of food security. These merits promote the concept of "nourishing urbanism" (Knight and Riggs 2010). Promoting urban agriculture enhances sustainable use of energy, water, land, and nutrients while increasing food production and access.

To be sustainable, there are numerous agronomic factors in urban agriculture which must be taken into consideration (Pearson et al. 2010). There are numerous constraints affecting yields of crops grown in urban ecosystems (Eriksen-Hamel and Danso 2010). Important among these are: low solar radiation, air pollution, shallow effective rooting depth, soil compaction, low soil fertility and elemental imbalance, soil pollution and contamination, frequent water imbalance (drought and inundation), and pests and pathogens. The adverse impacts of these constraints vary because of site-specific conditions. Yet, site-specific agronomic management can alleviate these constraints and enhance crop yields. Recommended management practices to alleviate these constraints include application of biosolids (i.e., compost, manure, mulch), use of balanced plant nutrients including micronutrients, supplemental irrigation, etc. Raise-bed, to increase the effective rooting depth, is an important option to improve crop yields in urban agriculture.

While soil contamination (with heavy metals such as Pb) is an important factor, there is little if any research on techniques to alleviate soil physical/hydrological constraints to enhancing productivity and quality of urban agriculture. Researchable priorities in this context include: identify management practices to alleviate soil compaction, enhance effective rooting depth, improve plant available water capacity in the root zone, increase water infiltration capacity, enhance soil aeration and gaseous exchange, improve bioturbation, and moderate soil temperature. There are also strong interactions between soil fertility and soil physical quality, especially in urban agriculture. Availability of both macro and micronutrients strongly influences the root system development and plant available water capacity. Indoor agriculture, using abandoned industrial and corporate buildings to raise fish and vegetables indoor, is also gaining momentum in large cities. The waste products are recycled in "aquaponic farming" to use as fertilizers. These are the futuristic urban farms in which nothing leaves the facility but the food. The world is running out of arable land to feed its growing population. Thus, urban agriculture is an important component of the food security equation.

1.8 Conclusions

Urban ecosystems, an expanding land use, strongly impact local, regional and global climate through alternations in energy budget, biogeochemical cycling of elements (C, N) and water, emissions of greenhouse gases, and alterations of landscape. Urban centers are major sources of gaseous emission with high radiative forcing and cause of air pollution. However, because of large and increasing area and the intensive management, the ecosystem C pool in urban landscape is anthropogenic driven rather than nature driven. The ecosystem C pool, both above-ground (biotic pool) and below-ground (pedologic pool) components, can be moderated through management. Land-based C sinks in urban ecosystems comprise of urban forests, home lawns and turfs, and urban agriculture. Judicious management (i.e., choice of species, fertilizer amount and rate, irrigation, mowing, pruning, mulching) is important to reducing the hidden C cost and increasing the net ecosystem C gains. Urban agriculture, enhancing multifunctionality of urban ecosystems, is gaining prominence. In addition to reducing food mileage, it advances food security, creates job/employment opportunities, and enhances social/cultural interaction. Judicious management of urban ecosystem is important and integral to a range of options to adapt and mitigate abrupt climate change. Similar to home lawns and recreational grounds, C foot print of urban agriculture must also be reduced through adaptation of sustainable practices. Alleviation of soil physical and nutritional constraints is important to sustainable production, and to making urban agriculture a net sink of atmospheric CO_2.

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Chapter 2
Adapting Urban Land Use in a Time of Climate Change; Optimising Future Land-Use Patterns to Decrease Flood Risks

Eveline van Leeuwen and Eric Koomen

Abstract It is increasingly acknowledged that a careful planning of urban areas is needed to cope with the negative effects of future climate changes. The planning process calls for finding a balance between various ecosystem services, such as, water and air purification, the regulation of rainfall, the preservation of natural and cultural values, increased flood risk, while at the same time providing ample space for societal demands in relation to residences, employment and recreation. In this paper we focus on the possibilities of adapting land in urban areas in such a way that possible negative effects of climate change, in particular flooding, are reduced as much as possible. We will illustrate this with a case-study in the Netherlands, using a GIS-based land-use simulation model in combination with a flood damage assessment module. This combination of tools is applied in a regional planning context to optimise land-use patterns taking into account flood risk and other water management issues.

Keywords Adaptation • Urban land-use • Flood risk • Damage assessment • GIS based land-use simulation

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List of Abbreviations

GHG	greenhouse gas
HIS	high water information system
IWM	integrated water management
UNDP	United Nations development plan

2.1 Introduction

Climate change will have different effects in different regions of the world. Different communities – depending on their socio-economic position and geographic location – may be affected differently. Nevertheless, it will gradually impact the lives of all people (van Leeuwen et al. 2009). It is increasingly acknowledged that a careful planning of urban areas is needed to cope with the negative effects of future climate changes. The planning process calls for finding a balance between various ecosystem services, such as, water and air purification, the regulation of rainfall, the preservation of natural and cultural values, increased flood risk, while at the same time providing ample space for societal demands in relation to residences, employment and recreation. In the UNDP Human Development Report 2007/2008, five key transmission mechanisms are distinguished through which climate change may impact on human development:

- Agricultural production and food security (through changes in rainfall, temperature and water availability).
- Water stress and water insecurity (through changes in run-off patterns and glacial melt).
- Rising sea levels and exposure to climate disasters (through accelerated ice sheet disintegration).
- Ecosystems and biodiversity (through world-wide quantitative and qualitative changes in ecological systems).
- Human health (through, e.g., heat waves and extreme summer and winter conditions).

In urban ecosystems, the impact of climate change, in particular related to human health is expected to be more severe compared to those living in rural areas. Heat waves are sporadic but recurrent, and are considered a mere annoyance, rather than a threat. However, the extra mortality because of the extremely warm summer in 2003 is estimated at 35,000 persons in Europe (AR4 2007). The effect of buildings is regarded as one of the main reasons for the urban heat island effect. Building masses increase the thermal capacity, which has a direct bearing on city temperatures. They reduce wind speed and radiate heat through the building fabric and air-conditioning equipment also tends to generate heat. The heat absorbed during the day by the buildings, roads and other constructions in an urban ecosystem is released

Table 2.1 Percentage of population living in urban areas

Year	World	More developed regions	Less developed regions	Europe	Netherlands
1950	29.1	52.5	18.0	51.2	56.1
1960	32.9	58.7	21.7	56.9	59.8
1970	36.0	64.6	25.3	62.8	61.7
1980	39.1	68.8	29.6	68.0	64.7
1990	43.0	71.2	35.1	70.5	68.7
2000	46.6	73.1	40.2	71.4	76.8
2010	50.6	75.0	45.3	72.6	82.9
2020	54.9	77.5	50.5	74.8	86.5
2030	59.7	80.6	56.0	77.8	88.6
2040	64.7	83.5	61.6	81.0	90.3
2050	69.6	86.0	67.0	83.8	91.8

Source: United Nations (2008)

after sunset, creating higher temperature differences between urban and rural areas (Rajagopalan et al. 2008). In urban areas, the impact of climate change, in particular related to human health, is expected to be more severe compared to more rural areas. Furthermore, millions of urban area residents are exposed to elevated levels of air pollutants that have been linked to adverse health effects and which can have synergistic effects when combined with heat (Semenza et al. 2008).

In addition, drought and floods claim more lives than all other natural catastrophes combined. However, as droughts mainly affect developing countries, floods have serious impacts all over the world. The ten cities with the highest population exposure today are almost equally split between developed and developing countries. When assets are taken into consideration, more developed countries enter the top 10, as the wealth of the cities becomes important (OECD 2007). The top 10 cities in this ranking are Miami, greater New York, New Orleans, Osaka-Kobe, Tokyo, Amsterdam, Rotterdam, Nagoya, Tampa-St Petersburg and Virginia Beach. These cities represent 60% of the total exposure, but are part of only three (wealthy) countries: USA, Japan and the Netherlands. However, in the future it is expected that both population and assets will increase significantly in developing countries, increasing the risk there as well (see Table 2.1).

This chapter focuses on the possibilities of managing land in peri- urban areas in such a way that potential negative effects of climate change, in particular flooding, are drastically reduced. This is illustrated by a case-study of a region in the province of Overijssel, the Netherlands. The study uses a GIS based simulation tool in a regional planning context to optimise land-use patterns taking into account future flood risks. The chapter also elaborates the relationships between urbanisation, green spaces and water management, while describing the methodology used to assess future flood risks. The chapter outlines expected flood risks and the optimised urban land-use patterns for a safe water management.

Fig. 2.1 An example of a dike structure along the river Rhine that consists of a smaller summer dike and the higher winter dike (to the *left*) (Photo by W.S. van Leeuwen)

2.2 Urbanisation, Green Spaces and Water Management

Global climate changes induced by increases in greenhouse gas (GHG) concentrations is likely to increase temperatures, change precipitation patterns and probably raise the frequency of extreme events (IPCC 2001). These changes may have serious impacts on society, in particular on river deltas because of both sea level rise and an increased occurrence of flooding events (Jacobs et al. 2000). Flooding events may cause enormous economical, social and environmental damage and even loss of lives (Booij 2005).

Adaptation to climate change by water management can be assessed in many different ways. One important aspect, however, is flood protection. Flood protection structures can be either artificial, such as dykes and dams, or natural, such as retention areas (see Fig. 2.1). Besides the artificial protection structures, natural structures are becoming increasingly important. Basic measures consist primarily of actions aimed at creating more space for the river and lowering high water levels, such as deepening river forelands, relocating dykes further inland, lowering the obstacles in the rivers, creating flood channels and enlarging summer beds. Apart from these basic measures, temporary measures can also contribute to water safety. An example is the retention areas. In the UK, recent reforms of rural and agricultural policy and a new strategic assessment of policy for flood risk management are leading to new approaches to flood risk management in rural areas in the future (Defra 2004).

In this perspective, green spaces are indispensable in moderating the impact of the negative consequences of climate change. Apart from the temporary storage of water, (urban) green is also important for the ample, absorption of pollutants and the release of oxygen. Furthermore, they maintain a certain degree of humidity in the atmosphere; moderate rainfall, influence changes in temperature and curb soil erosion, all contributing to a healthier urban climate for both mankind and nature. Green spaces form the basis for the conservation of fauna and flora in and around cities, and contribute to the maintenance of a healthy urban environment by providing clean air, water and soil thus improving the urban climate (van Leeuwen et al. 2010).

In particular in the future, when climate is likely to change in urban areas around the world, maintaining the balance of the city's natural urban environment will be of utmost importance. When urban green spots can be linked to more rural green areas, the effect can be even more significant. This renewed awareness of the value of nature, culture and landscapes in climate adaptation and mitigation is encouraging the conservation of these elements.

However, the conservation of urban green is not an easy task and requires strong planning policies. A possibility is to link it to water management planning. If well planned, integrated water management (IWM) can provide benefits in many ways. It can increase the space available for species, create opportunities for the development of new nature areas and contribute to expanding the ecological corridors thus enabling the migration of species. Indirectly, IWM can help to counteract flooding and drought and thus can prevent damages to the ecosystem. An important task of IWM and spatial planning is to protect existing and potential runoff and storage areas, to keep areas subject to the risk of flooding free of undesirable developments and to integrate water streams in urban development (van Gelder 1999). An important issue in this respect is whether the construction of buildings should be allowed in flood plains and how they should be protected. In addition, the timing of measures is very important, as are the consequences of the irreversibility of certain developments. Related to the costs and benefits of flood protection the questions include how can flood protection be combined with other activities in the most efficient and sustainable way. However, insights are generally lacking in the costs involved in floods, related *inter alia* to the indirect costs of floods and the ways in which businesses perceive these costs. Related questions in this respect are what is an acceptable flood risk and how, for example, does it relate to acceptable traffic risks.

2.3 Assessing Future Flood Risk

Risk is generally considered to consist of both the probability and the consequences of a failure (Howard 2002). To estimate the risk of flooding from an economic point of view, both the probability of a flood as well as the economic damage caused by a flood should be taken into account. This means that, for example, an agricultural area with low economic value and a high flood risk can have a lower economic flood risk than an area with more high value properties and a low chance of getting flooded. In this perspective, the economic flooding risk of areas such as the greater New-York area and the Amsterdam-Rotterdam region is enormous.

Future risk can be defined as the probability of a flood under changed scenarios multiplied by the consequence under changed scenarios. Although this definition is often used, it does not take into account two important aspects. One, the damage depends on the specific location of the dyke failure. Two, the total risk is not only the risk in a single year, but also the probability of dyke failure in the coming years plays a role. Not the heightened probability, but the expected yearly loss by flooding is the key variable in an optimal safety strategy in the case of economic growth (Eijgenraam 2006). Furthermore, the risk level can be based either on individual risk (the probability of being killed by a flood) and group risk (the probability of a group of people being killed by a flood) (Vrijling et al. 1998).

In the Netherlands, a country surrounded by sea and river water, elaborate models exist to calculate current flood risk (e.g. the High water Information System damage module described by Kok et al. 2005; Messner et al. 2007). However, more simple methods need to be applied when dealing with assessment of flood risk in the future due to the large degree of uncertainty involved. For the analysis described in this chapter, a straightforward approach to describe flooding probabilities and their impacts is proposed. This methodology uses simulated land-use patterns as a proxy for the potential economic value at risk in the future. This method has the advantage of being both robust and flexible enough to be applied in a strategic planning context. It does not aim to replace more elaborate methods, but it is meant to provide a quick overview of potential damages for various planning alternatives. The proposed approach relies on two components: (i) a land-use model to simulate future land-use patterns, and (ii) a simplified damage module that assesses potential flood damages based on the input parameters land use and inundation depth. Additionally, an estimate of the potential number of casualties is provided. The assessment of potential economic damage and number of casualties are based on relatively simple depth-impact functions. The latter are used that quantify the relationship between inundation depth and damage or casualties for different land-use types.

2.3.1 Simulating Future Land-Use Patterns

In order to simulate future land-use patterns, a spatially explicit land-use model refereed to as *Land Use Scanner* was used. This is a GIS-based land-use model that simulates future land use based on the integration of sector specific inputs from dedicated models (Dekkers and Koomen 2007; Hilferink and Rietveld 1999). The model offers an integrated view on all types of land use, dealing with urban, natural and agricultural functions. It is grid-based and this application uses a 100 × 100 m grid, to cover the terrestrial Netherlands in about 3.3 million cells. This resolution comes close to the actual size of building blocks and allows for the use of homogenous cells that only describe the dominant land use.

The model is based on demand-supply interaction for land, with sectors competing for allocation within suitability and policy constraints. The model employs a mathematical optimisation approach to allocate land to its optimal location given a set of boundary conditions related to the regional demand for land and the local suitability definition. The solution of this discrete allocation problem is considered optimal when the sum of all suitability values corresponding to the allocated land use is maximal. This allocation is constrained by two conditions: the overall demand for the land-use functions that is given in the initial claims and the total amount of land that is available for each function. The suitability maps are generated for all different land-use types based on location characteristics of the grid cells in terms of physical properties, operative policies and expected relations to nearby land-use functions. This suitability can be interpreted to represent the net benefits (benefits minus costs) of a particular land-use type in a particular cell. The higher the benefits (suitability) for that land-use type, the higher the probability that the cell will be used for that type. The economic rationale that motivates this choice behaviour resembles the actual functioning of the land market.

Unlike many other land-use models, the objective of the *Land Use Scanner* is not to forecast the dimension of land-use change but rather to integrate and allocate future land-use claims from different sector-specific models. The land-use simulations integrate expert knowledge from various research institutes and disciplines and thus represent the best-educated guess regarding the possible spatial patterns. It should be noted, however, that the simulations are based on many assumptions about future developments. They can by no means be seen as exact predictions and should therefore not be treated like that. For a more detailed description of the most recent model version and the calibration and validation of its new allocation algorithm the reader are referred to another publication (Koomen et al. 2008b; Loonen and Koomen 2009).

As the Stern (2006) report has observed, one of the main problems that policy makers face is that by the time climate change is apparent with a higher level of certainty, it will be too late to reverse its effects or to effectively undertake adaptation. Therefore, often scenarios are developed to picture a possible future situation.

Scenarios have become a respected and powerful tool in modern policy analysis, in both private and public domains. They are particularly useful in decision and policy situations that are characterized by a high degree of future uncertainty (with often structural or systematic changes in the context or in the functioning of a relevant policy system). Instead of deterministic, stochastic or blueprint planning techniques for short- to medium-term policy issues, scenario's are operational tools for complex decision making that is marked by long-term and largely unpredictable uncertainty where visioning on future developments is necessary. Scenario analysis is then a knowledge-based rational tool to explore uncertain futures, with the aim to respond properly to future challenges. It is a learning tool ('flight simulator') that serves to arrive at informed and accountable decisions and/or solution trajectories. It does not aim to identify the best possible future, but to design a rational and transparent mechanism for coping with uncertain futures (van Leeuwen and Nijkamp 2007).

In the system of the Land-use scanner, land-use simulations are normally executed following different socioeconomic scenarios that are made spatially explicit with the help of several sector-specific models and a number of additional assumptions. Various specialized institutes contribute to this development of socioeconomic scenarios.[1] Land-use simulation normally starts by creating a 2010 land-use map from a 2000 base map taken from Statistics Netherlands (CBS 2002). In this step current, explicit land-use plans, mainly taken from the new map of the Netherlands survey (NIROV 2005), are included in the simulation to represent autonomous developments. Based on this base situation, simulations for 2040 are made for different scenarios according to the scenario-specific assumptions and sector-related developments.

2.3.2 Estimating Damage

Within the land-use scanner there is a module that especially focuses on damage: the Damage Scanner. Per scenario, per strategy and per year (2015, 2040), the flood effects are computed by applying damage functions to maps describing water levels after flooding and land-use configurations. The main elements of the Damage Scanner necessary to take the various steps performed in the damage calculations are shown in also Fig. 2.2. The calculation of the number of casualties is done in a similar way and not explicitly described here.

First of all, the required land-use maps are produced by the land-use model itself, each reflecting the scenario specific assumptions. Additionally, the current model configuration also contains several inundation maps reflecting different sea-level rise scenarios (25, 60, 80, 150 and 300 cm) in combination with two adaptation strategies. Furthermore, the inundation map representing the initial situation is a collection of available inundation simulations that were collected and integrated by the provinces.[2]

To assess potential flood damages the information of the land-use maps and the inundation maps are combined by using simple depth-damage functions. These functions provide the maximum damage values (total loss) for 14 different land-use classes and the respective growth of the function, which quantifies the relationship between inundation depth and damage. These functions are derived from the High water Information System (HIS), which is the standard software tool in the Netherlands to evaluate flood damages.[3]

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[1] The Netherlands Bureau for Economic Policy Analysis (CPB) and the ABF Research company have provided the expected amount of residential development (ABF 2006; CPB et al. 2006). CPB has delivered the demand for industrial and commercial land and office space (CPB 2002; CPB et al. 2006) and the Agricultural Economics Research Institute (LEI) the projections for agricultural land-use changes (Helming 2005).

[2] Together with other risk maps the flood risk map is available from the portal: www.risicokaart.nl.

[3] For more details on the HIS damage module and the procedure deriving the damage functions for the Damage Scanner, the reader is referred to other sources (Huizinga et al. 2004; Klijn et al. 2007; Van der Hoeven et al. 2009).

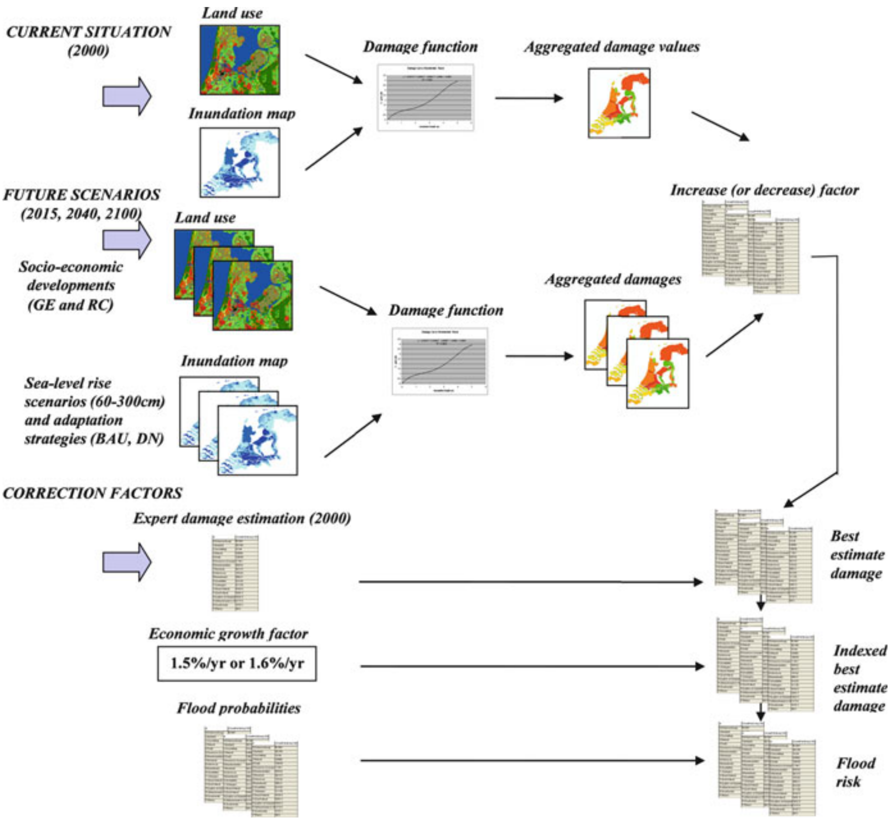

Fig. 2.2 Flowchart of the flood risk assessment with the damage scanner approach

The different adaptation strategies are referred to as 'Do Nothing Strategy' and 'Business as Usual Strategy' and describe two contrary alternatives how to react to a rising sea level. The assumption behind the 'Do Nothing Strategy' is that dikes heights remain unchanged. This leaves the water level in the dike rings unchanged but leads to increasing flooding probabilities. In contrast, the 'Business as Usual Strategy' assumes that dikes are raised in line with the sea level rise what results in higher water tables within the dike rings and thus to higher damage potentials. The probability of a flood event remains unchanged following this adaptation strategy. Figure 2.3 provides an example of the damage function for the land-use class 'Residential – Low Density', which has a maximum damage value of four Million €/ha.

Since the Damage Scanner gives a rough overview only, it is used to calculate the increase in damages according to a certain scenario compared to the calculated damage potential for the year 2000. The thus derived increase factor (or decrease) is subsequently multiplied with the best estimate currently available for each dike ring.

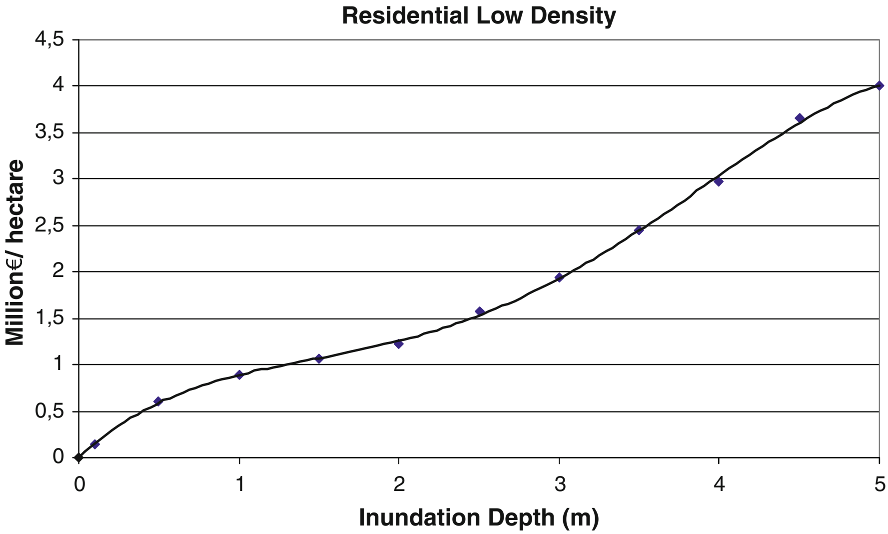

Fig. 2.3 Example of a damage function showing the relation between water depth and economic damage for the high density residential land-use type

These figures were initially derived by Klijn et al. (2004) (see also Klijn et al. 2007) and shall represent the most realistic estimation of potential flood damages for each dike ring. By multiplying the increase factor with the best estimate, possible over- or underestimations in the damage calculation resulting from unrealistic inundation maps can be corrected. The resulting damage figures are subsequently multiplied with an economic factor to represent growth in wealth according to scenario-specific conditions. In a recent study, this approach was applied to assess the development of exposure to large floods in the Netherlands during the 1900–2100 period (de Moel et al. 2011).

2.4 Optimising Urban Land-Use Patterns in the Regional Planning Context of Overijssel

The Netherlands is a ‘decentralised unitary state’ with a three-tier system of government (see Fig. 2.4). The country is split up into 12 provinces and around 450 municipalities. All three-tiers of government have planning powers. National and provincial land use plans are broad framework plans and policy guidelines. The municipalities have the statutory power to make both framework plans, as well as binding land allocation plans. The Spatial Planning Act includes a consistency requirement for local and regional plans and for the plans of the spending departments. These plans must comply with the national framework plans for spatial planning (van der Valk 2002).

Fig. 2.4 The location of the Netherlands and its main rivers and provinces

The new national Spatial Planning Act introduced in 2008 calls for a more pro-active role of Dutch provinces in the policy arena. In fact, it demands the definition of Regional Spatial Strategies that are new in process, content and legal status. These Strategies are the main guiding documents in spatial planning at the regional and local level and typically focus on the year 2020 with a further outlook on additional developments until 2040. To draft such Strategies two questions need to be answered: what regional spatial developments can be expected in the future?; and what role can regional policy making play to direct such developments? Land-use simulation can help solve these questions.

In 2008, a number of provinces defined their Regional Spatial Strategies by creating an outlook on possible spatial developments until 2040 using land-use simulations (Atzema et al. 2008; Koomen et al. 2008a; Kuijpers-Linde et al. 2008). In this chapter specifically focuses on the case of Overijssel (see Fig. 2.5) where land-use simulations

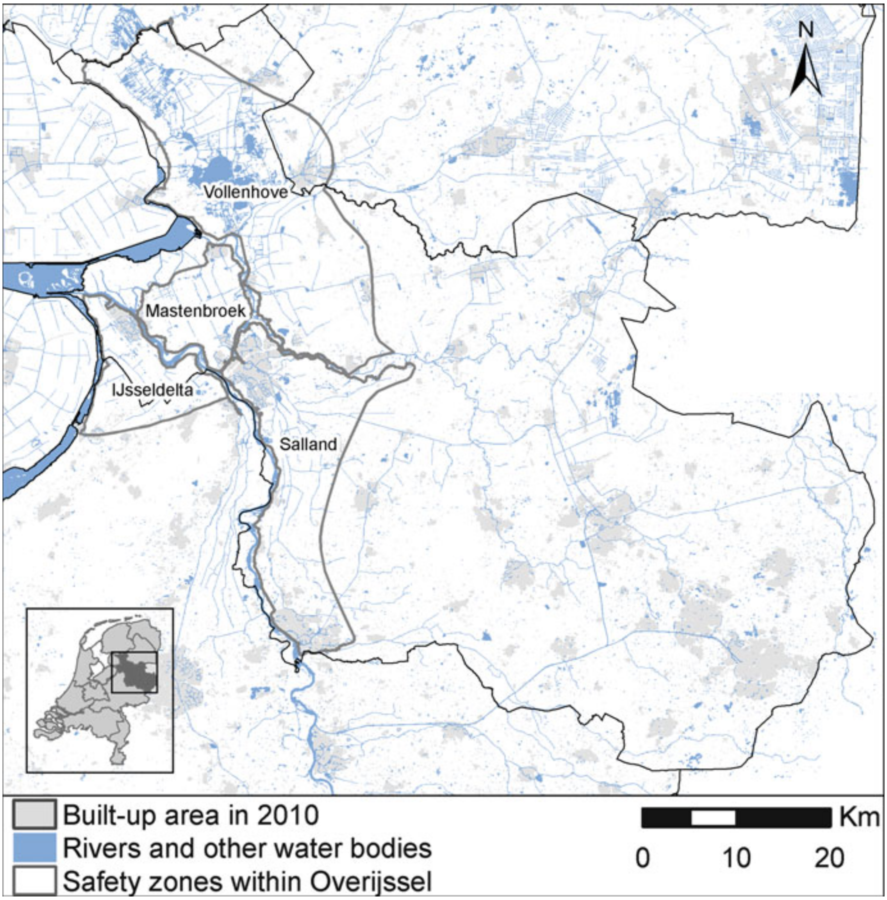

Fig. 2.5 The province of Overijssel and its safety zones for flood

were used to depict potential autonomous spatial developments and to simulate possible spatial implications of new policy alternatives. These studies not only showed the task at hand for these provinces, but also explored possible solutions for these tasks. Overijssel decided to adjust their prevailing strategic plans for the physical environment in 2008. Such a new strategic plan was especially necessary in view of several new themes on the policy agenda such as climate change and new insights in demographic developments. Climate change refers in this case foremost to sea-level rise, increased river discharge and more frequent excessive rainfall causing a series of water management issues. The introduction of the new national spatial planning act provided a good opportunity to redefine the provincial aims and embed these in spatial plans, finding a new balance between urban areas and green space. For this reason the provincial council decided to develop a new and integrated strategy for the physical environment that replaced five individual plans and functions: the spatial structural vision; the regional water management plan; the environmental policy plan; the provincial transport plan; and the soil plan.

The policy formulation process of the province of Overijssel was assisted with the Land Use Scanner model in three subsequent stages:

– providing trend-based spatial developments and their potential consequences in relation to, for example, flood risk and increased urbanisation in valuable landscapes;
– depicting potential spatial policy alternatives following coherent sets of spatially explicit ambitions;
– assessing the potential spatial developments and related sustainability impacts of the preferred policy alternative.

The latter stage was performed as part of the Strategic Environmental Assessment that was required before the new Spatial Strategy could be implemented. This process is described in more detail elsewhere (Koomen et al. 2010), and the remainder of this paper focuses on the first two, more exploratory stages of the planning process.

2.4.1 Providing Trend-Based Spatial Developments

To set the scene for the policy development process two trend-based depictions of future land use were created. These helped answer questions such as: to what degree are current spatial developments 'climate proof', and what are the impacts of possible spatial policies? From the national Second Sustainability Outlook study carried out by the Netherlands Environmental Assessment Agency (PBL 2010) two trend-based simulations of future land use were selected that aim to show probable spatial developments based on past developments and current (local and national) plans and policies. These simulations follow the same (spatial) policies but differ in the amount of urban change that is expected: following a moderate and high growth scenario respectively. These growth rates correspond to the Transatlantic Market and Global Economy scenario described in the Welfare, Prosperity and Quality of the living environment study of the Dutch assessment agencies (CPB et al. 2006).

In close cooperation with various representatives of the province, the national simulations were adjusted to better match current regional spatial policies. This feedback was organised through a number of workshops in which provincial policy makers reacted on intermediate results. Their local knowledge allowed an iterative fine-tuning of the simulation, for example, adjusting the importance of certain national policies and limiting urban growth around smaller towns and villages.

Based on these regionalised trend-based simulations of future land use a number of spatially-explicit assessments was made to highlight potential impacts on several spatial policy themes: built-up developments in the National Ecological Network, large-scale developments in the National Landscapes, building in flood-prone areas. These analyses were initially performed following a straightforward pixel-by-pixel

Table 2.2 Flood frequency and potential damage per safety zone in current situation (2000) and 2040 according to a trend-based simulation and a policy alternative

Safety zone	Size [ha]	Flood frequency	Potential damage current situation [M€]	Potential damage trend-based scenario [M€]	Potential damage policy alternative [M€]
Vollenhove	50,645	1/1,250	2,650	5,305	5,180
Mastenbroek	9,546	1/2,000	1,500	2,288	2,389
IJsseldelta	11,651	1/2,000	1,200	2,078	2,026
Salland	28,162	1/1,250	5,400	7,251	6,943
Total			10,750	16,922	16,537

Note that potential future damage is expressed in year 2000 prices (thus excluding the 1.5% economic growth factor discussed in Sect. 3.2) to allow for a clearer comparison with the potential damage in 2000 (Adapted from: Province of Overijssel 2009)

comparison between the urban (residential, commercial and greenhouse) locations in the maps showing current and simulated land use. The number of new urban locations within the sensitive policy zones was mapped. To assess the potential damage incurred by flooding the damage estimation method was applied as described in a preceding section.

Table 2.2 list the results of this calculation. It shows the four safety zones within the province that are situated in the flood plain of the major river IJssel (a branch of the river Rhine). The table lists for each zone its size and the expected flood frequency, which is basically the design safety level; the protective dikes are designed in such a way that they can withstand a flood with this return period. Furthermore, it shows potential damages expressed in Million euro for both the current situation (2000) and the two potential future situations. These future simulations were made with the Land Use Scanner model and describe the year 2040 following a trend-based scenario and a policy alternative that was proposed within the framework of the new Regional Spatial Strategy. Both simulations of future land use follow the same assumptions related to a moderate growth of urban areas but they differ in terms of governmental actions.

Table 2.2 clearly shows that potential flood-induced damages are likely to increase. This is partly due to expected land-use changes – notably urbanisation - within the zones, and can partly be attributed to an expected rise in sea level until 2040 of 25 cm. Based on this and other impact assessments several spatial policy alternatives were defined as is discussed below. One of these policy alternatives is also included in the table to indicate the difference in potential damage with the trend-based alternative. This difference is relatively small as the increase in potential damage is to a large extent related to impact of sea-level rise on currently existing urban areas. Nevertheless the policy alternative leads to a small decrease in potential damage in the two zones with the highest frequency and potential damage (these zones are Vollenhove and Salland), due to adaption policies that enforce a shift in the location of new urban areas.

2.4.2 Depicting Spatial Policy Alternatives

As part of the policy formulation process, the province of Overijssel wanted to explore different options for new spatial policies. To structure this exploratory process several coherent sets of policy ambitions were defined that were translated into corresponding maps of future land use, using the Land Use Scanner model. These maps thus show which spatial developments can be expected when certain policy ambitions are taken as the leading objective in spatial planning. The province was interested in the potential spatial developments related to three different coherent sets of spatial policy ambitions related to the themes of: (1) water management; (2) safety and health; and (3) compact urbanisation. These policy ambitions were made operational by specifying spatially explicit restrictive zoning regulations. These restrictions were applied within the high spatial pressure scenario as this highlights the impact of the policy measures more clearly. As such, it prepares policy makers for a worst-case scenario with respect to the spatial developments that have to be accommodated in the policy alternatives. To illustrate the process of depicting the spatial policy alternatives are briefly described in relation to the construction of the water management related alternative.

During several interactive sessions with provincial policy makers the key characteristics of this policy alternative were defined, their implementation in the model was discussed and preliminary simulation results were evaluated. This process led to the addition of many different datasets describing the stimulating or restricting policies associated with the policy alternative to create suitability maps for the urban land-use types that differ from the other policy alternatives. Several different water management related policies are included here. To prevent drought, groundwater protection areas are kept free from urban development. To limit the potential impact of flooding, urban development is not allowed within flood-prone areas near the major waterways, or in areas with high inundation risks, such as river flood plains. An overview of the implemented restrictions is provided in Fig. 2.6. Figure 2.7 shows the development of this specific policy alternative and compares its distribution of urban areas with the trend-based scenario. This comparison indicates a shift in developments from the lower lying cities in the west of the province to cities and villages situated on slightly higher grounds in the north and east of the province.

As the alternatives are not defined in isolation, they share the same demand for land and have a common basis for the definition of suitable locations. This approach secures the option to obtain a meaningful comparison between the alternatives. The policy alternatives emphasise different policy objectives and show the potential resulting developments to an extent that is not exactly likely to happen in reality. This over emphasis has the advantage of presenting a clear image of potential positive or negative impacts of proposed policies. By comparing the simulation results of a trend-based scenario and the theme-specific policy alternatives spatial differences can be visualised.

The province used the spatial exploration to frame the possible effects of new building developments on the National Ecological Network, preservation of the

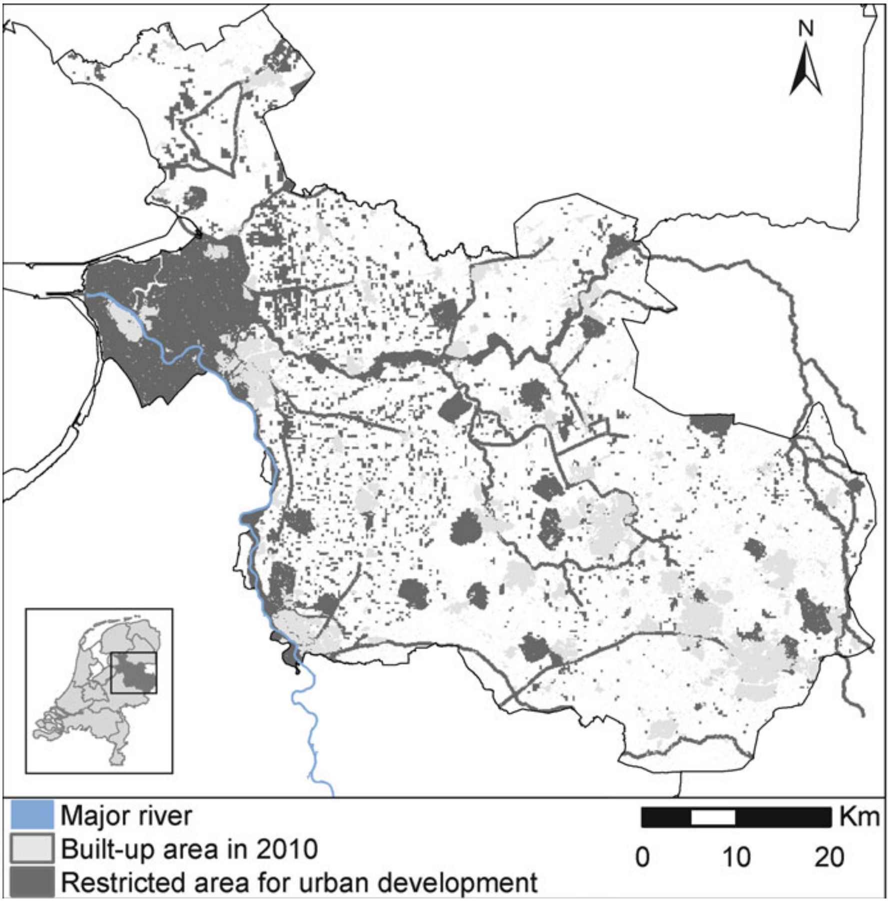

Fig. 2.6 The water management related restrictions applied in the simulation of this policy alternative

National Landscapes and flood risks. The study provided the province with insights that were used in the policy formulation process for the Regional Spatial Strategy. In combination with additional research, the simulation results of the spatial exploration study initiated the incorporation of specific spatial policies in the legal documents related to the new Regional Spatial Strategy. This resulted in a more explicit delineation of the zoning regulations that aim to protect landscapes (National Landscapes) and biodiversity (National Ecological Network). Additionally, the simulation results helped delineate specific areas that specify where water storage is deemed necessary in times of excessive rainfall or flooding or where safety regulations apply that limit the development of specific types of urban land use. Special regulations were also drafted for the areas that face an increased risk of river flooding.

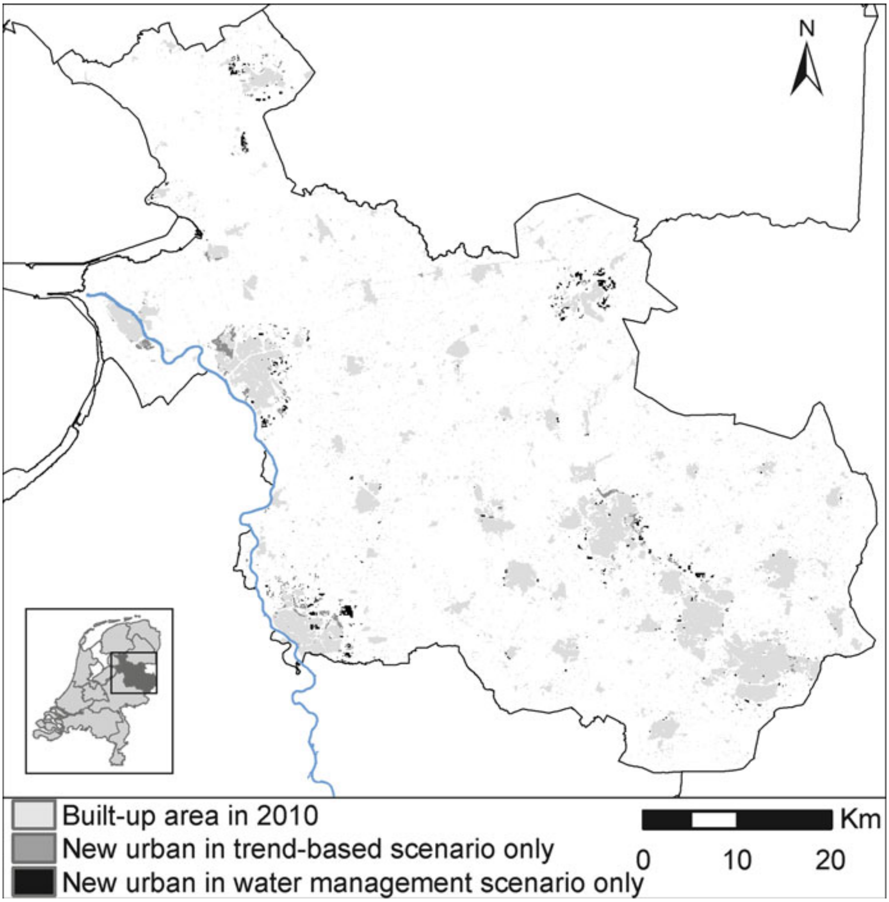

Fig. 2.7 The relocation of built-up area as compared to a trend-based scenario; *black* indicates new urban areas according to the water management alternative, these replace the *dark grey* urban areas that were initially simulated in the trend-based scenario. Note that new urban areas according to both scenarios are not shown here for visual clarity

2.5 Conclusions

In urban areas, the impact of climate change, in particular related to human health is expected to be more severe compared to more rural areas. The effect of buildings is regarded as one of the main reasons for the urban heat island effect. Furthermore, millions of urban area residents are exposed to increased levels of urban air pollutants that have been linked to adverse health effects and which can have synergistic effects when combined with heat. In addition, drought and floods claim more lives than all other natural catastrophes combined. However, as droughts mainly affect developing countries, floods have serious impacts all over the world, not the least in the Netherlands.

In order to prevent regions from flooding, integrated water management is seen as an important tool in which several different water management related policies are included with spatial planning. To prevent drought, groundwater protection areas are kept free from urban development. To limit the potential impact of flooding, urban development is not allowed within flood-prone areas near the major waterways, or in areas with high inundation risks, such as river flood plains. Furthermore, water retention areas are appointed in which blue and green enforce each other. In these areas, rivers get more space, groundwater is protected against pollution of human activities and flora and fauna are free to develop. This results in high quality green areas, often partly accessible for urban residents, which decreases flood risk downstream. A clear example of exploiting eco-system services.

This chapter describes a method to assess future flood risk using three main components: a land-use model to simulate future land-use patterns, a scenario framework to describe potential future conditions and an assessment tool that links simulated land-use patterns with potential economic damage.

In order to simulate future land-use patterns we used a spatially explicit land-use model refereed to as *Land Use Scanner*. Based on regionalised trend-based simulations of future land use a number of spatially-explicit assessments was made to highlight potential impacts of climate change on several spatial policy themes: built-up developments in the National Ecological Network, large-scale developments in the National Landscapes and building in flood-prone areas.

The present assessment focuses mostly on the most prominent impact of climate change; increased flood risk. To estimate these impacts a so-called 'Damage scanner' has been developed within the *Land Use Scanner*. This instrument computes the economic damage and number of casualties per scenario and strategy. Per scenario, per strategy and per year (2015, 2040), the flood effects were computed by applying damage functions to maps describing water levels after flooding and land-use configurations.

The results of the damage scanner show that potential flood-induced damages are likely to increase. This trend is partly due to expected land-use changes – notably urbanisation – within the zones, and can partly be attributed to an expected rise in sea level until 2040. However, the difference between the trend scenario and the policy scenario appears to be relatively small. This is because most potential damage is related to the impact of sea-level rise on currently existing urban areas. Nevertheless, the policy alternative is expected to lead to a small decrease in potential damage due to adaption policies that enforce a shift in the location of new urban areas.

Although the effect of specific policies expressed in monetary terms is relatively small, together with additional research, the simulation results of the spatial exploration study initiated the incorporation of specific spatial policies in the legal documents related to the new Regional Spatial Strategy. In particular the link between red, green and blue (i.e. built-up areas, green areas and water) is considered as very important by the province of Overijssel. The province aims at integrating and combining water storage areas with relative capital extensive spatial functions such as recreation, agriculture, green provision for urban residents and nature conservation. An important synergy is expected to appear between (urban) quality of life, green provision, water safety and water quality. By constraining new building activities in part of the retention areas, the safety of larger urban areas downstream is supposed to increase. Furthermore, both the quality of ecological systems, as well as ground-water systems are expected to increase.

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Chapter 3
Comparison of Methods for Estimating Carbon Dioxide Storage by Sacramento's Urban Forest

Elena Aguaron and E. Gregory McPherson

Abstract Given the increasing demand for carbon dioxide storage estimates in urban areas and the high cost for ground-based inventories, there is need for more efficient approaches. Limited open-grown urban tree species biomass equations have necessitated use of forest-derived equations with diverse conclusions on the accuracy of these equations to estimate urban biomass and carbon storage. Our goal was to determine and explain variability among estimates of CO_2 storage from four sets of allometric equations for the same ground sample of 640 trees. Also, we compare the variability found in CO_2 stored and sequestered per hectare among estimation approaches for Sacramento's urban forest with the variation found among six other cities. We found substantial variability among the four approaches. Storage estimates differed by a maximum of 29% and ranged from 38 to 49 t/ha. The two sequestration estimates differed by 55%, ranging from 1.8 to 2.8 t/ha. To put these numbers in perspective, they amounted to about one-tenth and one-quarter of the maximum differences in CO_2 storage and sequestration rates among six cities, respectively. i-Tree Eco produced the lowest storage estimates, perhaps because it relied exclusively on forest-based equations and applied a 0.80 correction factor to open-grown trees. The storage estimates produced by i-Tree Streets and CUFR Tree Carbon Calculator (CTCC) were the highest, while Urban General Equations produced relatively low estimates of CO_2 storage. Eco produced lower estimates of CO_2 sequestration rates than the CTCC across a range of species. Eco's reductions for tree condition and projected mortality may partially explain the difference. An analysis of the roles of tree growth modeling and biomass equation selection for a green ash tree illustrated how the dynamic interaction between tree growth and biomass

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storage rate can influence the temporal stream of sequestration in complex ways. Based on these results we conclude that applying UGEs to remotely sensed data that accurately classify broadleaf, conifer and palm tree types in the Sacramento region is likely to produce conservative results compared to results from urban-based species-specific equations. The robustness of this result needs to be tested with different tree populations, and research is needed to establish relations between remotely-sensed tree crown projection area and dbh values required for biomass calculation. Of course, ground-based inventories remain necessary for more accurate estimates of CO_2 storage and for municipal forest management and health monitoring purposes.

Keywords Carbon storage • Sequestration rates • Allometric equations

List of Abbreviations

BVOCs	Biogenic volatile organic compounds
CLE	Crown light exposure
CTCC CUFR	Tree Carbon Calculator
CUFR	Center for Urban Forest Research
STRATUM	Street Tree Resource Assessment Tool for Urban forest Managers
SUFES	Sacramento Urban Forest Ecosystem study
UFORE	Urban Forest Effects Model
UGEs	Urban general equations

3.1 Introduction

Growing concern about climate change has led to research quantifying the effects of urban forests on atmospheric carbon dioxide (CO_2) (Nowak 1994; McPherson 1998; Jo 2002; Nowak and Crane 2002; Pataki et al. 2006; Escobedo et al. 2010; Stoffberg et al. 2010; Zhao et al. 2010). Most of these studies have found that urban forests can be important carbon sinks, although there is a general lack of information on urban tree biomass allometry. Similarly, relatively little is known about the release of CO_2 into the atmosphere from combustion of fuels used to power equipment and vehicles during planting and tree care activities. Once dead, trees release most of the CO_2 they accumulated through decomposition. The rate of release depends on how the wood is utilized.

A number of computer tools have been developed to calculate carbon storage and sequestration rates of urban trees, as well as emission reductions from power plants as a result of building heating and cooling energy savings. These tools produce estimates of atmospheric CO_2 reductions from urban forests that are used for policy, management, and educational purposes. To better understand the variability associated with using different tools to estimate CO_2 storage, this paper examines differences among estimates produced by three different tools and three urban general equations (one for broadleaves, one for conifers and one for palms) for the same 640 ground-sampled trees in Sacramento.

3.1.1 Carbon Storage

As part of their biophysical processes trees capture and release CO_2 to the atmosphere. During photosynthesis leaves absorb CO_2 through the stomata and, using the energy from the sun, convert it into oxygen, carbohydrates and water that are then used in the production of wood structures as well as vitamins, resins and hormones needed for growth and tree health. Trees obtain energy to grow from the carbohydrates synthesized during photosynthesis, and they respire by releasing CO_2, water, and heat energy. The combined effect of photosynthesis and respiration results in net storage of CO_2 by the tree.

The term "carbon dioxide storage" refers to the accumulation of woody biomass as trees grow over time. The amount of CO_2 stored at any one time by urban trees is proportional to their biomass and influenced by tree density and management practices (McPherson 1994).

"Carbon dioxide sequestration" refers to the annual rate of storage of CO_2 in biomass over the course of one growing season. Sequestration depends on tree growth and mortality, which in turn depends on species composition and age structure of the urban forest (McPherson 1998).

"Carbon stock" is the stored carbon in one place at a given time. Forest carbon stocks include living and standing dead vegetation, woody debris and litter, organic matter in the soil, and harvested stocks such as wood for wood products and fuel (California Climate Action Registry 2008).

3.1.2 Allometric Equations

Estimates of carbon storage are obtained from allometric equations that use several parameters to calculate tree biomass: diameter at breast height (dbh), tree height, wood density, moisture content, site index and tree condition. Parameters like wood density and moisture content vary not only among species but also among trees of the same species. Even within a single tree there can be significant differences in density and moisture content (Domec and Gartner 2002; RPBC 2003). Therefore, some error is associated with the use of average densities and moisture contents in allometric formulas.

There are two types of allometric biomass equations: volumetric and direct. Volumetric equations calculate the above ground volume of a tree using dbh and tree height for the species. Direct equations yield above ground dry weight of a tree using dbh and tree height.

The methodology to convert green volume into biomass and eventually to stored CO_2 is well established (Markwardt 1930; Markwardt and Wilson 1935; Forest Products Laboratory 1987; Hansen 1992; Simpson 1993; Jenkins et al. 2003a, b). Estimating biomass and CO_2 using volumetric equations is a process that entails calculating dryweight biomass, then carbon (C) and stored CO_2 equivalents (McPherson et al. 2008). Converting the fresh weight of green volume into dry-weight requires use of density conversion factors that were published by Markwardt and Wilson (1935). The biomass stored below ground is added to above ground biomass (total biomass = 1.28 * above ground biomass) (Husch et al. 1982; Tritton and Hornbeck 1982; Wenger 1984; Cairns et al. 1997). Wood volume (dryweight) is converted to carbon by multiplying by the constant 0.50 and carbon is converted to CO_2 by multiplying by 3.67 (molecular weight of carbon dioxide) (Lieth 1963; Whittaker and Likens 1973).

3.1.3 Urban-Based Allometric Biomass Equations

There are 26 species–specific equations for trees growing in open, urban conditions. Urban-based biomass equations were developed from street and park trees measured in California (Pillsbury et al. 1998) and Colorado cities (McHale et al. 2009). Two sets of biomass equations were published, one set based only on dbh where

$$(\mathrm{dbh}),\ \mathrm{biomass} = a * (\mathrm{dbh})^{b} \tag{3.1}$$

and the other set based on dbh and tree height where

$$\mathrm{biomass} = a * (\mathrm{dbh})^{b} * (\mathrm{height})^{c}. \tag{3.2}$$

Very limited destructive biomass sampling has been conducted on urban trees to verify the accuracy of estimates from these equations across a range of growing conditions. In addition, limited research has quantified differences in growth and biomass accumulation between open-grown and non open-grown trees. The magnitude of error associated with the frequent practice of applying forest-based equations derived from measurements on non-open grown trees to open-grown trees is an important research question.

3.1.4 Forest-Based and Urban-Based Equations

Biomass equations for open-grown urban trees should reflect how different growing conditions, stresses and management practices influence the partitioning of biomass to bole, branches, foliage and roots compared to forest trees. Although not well documented, carbon partitioning might be different for open-grown trees than for forest trees. Carbon partitioning for a typical forest tree was reported to be about

17% in roots, 50% in trunk, 30% in branches and stems, and 3% in foliage (Birdsey 1992). Forest trees often grow in denser stands and develop smaller crowns and longer trunks than open-grown trees.

Trees in open-grown conditions do not compete as directly with other trees, and are allowed to branch into spreading crowns that support ample foliage. The growth of open-grown trees is often enhanced by periodic irrigation and care, as well as elevated levels of carbon dioxide and nitrogen deposition. Some studies indicate that urban trees grow faster than forest trees and sequester more CO_2 on a per tree basis (Jo and McPherson 1995; Nowak and Crane 2002). However, urban trees have stressors, such as constricted space, poor soils, pests, and vandalism that can restrict their growth. Little research has been published on carbon partitioning for urban trees, but there is some evidence that they partition relatively more carbon in branches and foliage, and less carbon to the bole compared to forest trees (Xiao 1998; Brack 2002).

Based on aboveground biomass weighed for 30 removed trees in Oak Park, IL, Nowak (1994) found less biomass than predicted with forest biomass equations and inferred that the biomass for open-grown trees should be multiplied by a factor of 0.8 when a forest-based allometric equation was applied. However, McHale and others (2009) found that applying the 20% reduction to carbon estimates for the Fort Collin's street tree population resulted in an estimate that was 30% less than the urban-based predictions. They concluded that standard application of the 20% reduction may lead to conservative estimates of biomass.

3.1.5 General Equations

There is a great deal of uncertainty associated with the application of biomass equations across a population of trees in a city or urban region. Although 26 species-specific allometric equations have been developed for city trees, their accuracy has not been well established, especially when applied across a range of climates, growing conditions and tree sizes.

Tree species richness is high in cities. Frequently, there are over 100 species of trees in urban populations. Because of this diversity and the limited number of urban-based allometric equations, most species are assigned a forest-based biomass equation from the same or similar species, or they are assigned an urban-based equation from a similar species. The magnitude of error associated with species assignment depends on the proportion of population assigned, as well as goodness of fit in terms of matching actual biomass to biomass predicted by the allometric equations.

The development and application of generalized equations is one approach to resolving the high variability and uncertainty associated with application of these allometric equations in both urban and forested environments (Jenkins et al. 2003a, b; McHale et al. 2009). Forest-based general equations have been developed for hardwoods, softwoods, and other types of trees, but no general equations have been developed using urban-based biomass equations.

3.1.6 Carbon Storage Estimation Approaches

i-Tree is public-domain software developed by the USDA Forest Service and cooperators for urban forestry analysis and benefits assessment. i-Tree helps communities to strengthen their urban forest management and advocacy efforts by quantifying the structure of community trees and the ecosystem services they provide. Within i-Tree, carbon storage by entire urban forest tree populations is assessed using Eco (formerly UFORE) whereas storage by discrete street tree populations is assessed using Streets (formerly STRATUM).

i-Tree Eco quantifies urban forest structure, environmental effects, and value to communities from field data and local hourly air pollution and meteorological data (Nowak et al. 2008). Setting up Eco projects for small, complete populations of trees is relatively straightforward because no sampling is involved. Eco sampling projects are typically used where the designated study area is too large to cost-effectively inventory the entire tree population. Sampling projects obtain estimates of the characteristics and benefits of a study area from a series of pre-selected sample plots. Such projects usually require project setup that can include characterization of land use and random selection of plot locations in a city using aerial photography or GIS. Field data costs $200 to $400 per 0.04 ha plot when collected by contracted professionals (Maco June 11, 2008, personal communication). A typical regional study will cost approximately $80,000 for 300 plots. Volunteers can be trained to collect field data, but there are costs associated with training, supervision, data processing, and quality control.

i-Tree Streets is a street tree specific analysis tool for urban forest managers that uses tree inventory data to quantify structure, function and value of annual benefits (Maco and McPherson 2003; McPherson et al. 2005). Users have the option of analyzing an existing street tree inventory or completing a new Streets-compatible inventory (complete or sample).

Eco and Streets produce tables and charts of information on urban forest structure, function, and value that can be exported in a variety of formats. Both models calculate the value of ecosystem services: CO_2 storage and sequestration, building energy effects and reduced CO_2 emissions, air pollution removal and release of biogenic volatile organic compounds (BVOCs). Streets includes output on rainfall interception and property value increase.

The i-Tree programs require specific types and amounts of data to accurately project the structure and benefits of urban vegetation. The validity of results depends on how closely users adhere to project setup and sampling protocols. Although the i-Tree programs are user-friendly, there is not much opportunity to adjust inputs or modify the calculations. This “black-box” design limits usefulness of the programs for customized applications.

Developed by the USDA Forest Service and first released in 2008, the Center for Urban Forestry Research (CUFR) tree carbon calculator (CTCC) is a free Microsoft Excel spreadsheet that provides carbon-related information for a single tree in one of 16 U.S. climate zones. It is the only tool approved by the Urban Forest Project Protocol for quantifying CO_2 sequestration from tree planting projects (Climate Action Reserve 2010).

Tree size data are based on growth curves developed from samples of about 1,000 street and park trees representing approximately 20 predominant species in each of the 16 reference cities (Peper et al. 2001a, b). Most of the biomass equations and calculations used to derive total CO_2 stored, total stored above ground, and annual CO_2 sequestered are from open-grown urban trees. To determine effects of tree shade on building energy performance, over 12,000 simulations were conducted for each of the 16 reference cities using different combinations of tree sizes, locations, and building vintages (Simpson and McPherson 2000).

Users enter information for a single tree, such as its climate zone, species name, size or age. The program estimates how much CO_2 the tree sequestered in the past year and over its lifetime. It calculates the biomass (dry weight) that would be obtained if it were removed. Trees planted near buildings to reduce heating and cooling costs require additional inputs because they also reduce GHGs emitted by power plants while generating electricity. These inputs include information on the tree's distance and compass bearing relative to a building, building vintage (its age, which influences energy use), and types of heating and cooling equipment. The CTCC automatically calculates annual heating and cooling energy savings, as well as associated power plant reductions using existing or user supplied emission factors for local utilities.

Another approach for calculating CO_2 storage in city trees utilizes existing imagery obtained by remote sensing with urban general equations (UGEs) for broadleaf, conifer, and palm tree types. Remotely sensed imagery is becoming increasingly available at higher resolutions and lower cost. In many cases, imagery exists for tax assessment and planning purposes. Many communities are conducting tree canopy cover assessments. It is estimated that the cost for such an assessment using available high resolution imagery (e.g., IKONOS, Quickbird) ranges from \$0.15 to \$0.25 per ha. The cost for a typical assessment for a 100,000 ha region will be approximately \$20,000.

Estimating CO_2 storage in urban forests with remote sensing and UGEs may be less expensive than ground-based sampling, but almost certainly will be less accurate. The accuracy of tree canopy cover classification typically ranges from 78% to 90% (Schreuder et al. 2003; Baller and Wilson 2008). Xiao and others (2004) reported mapping urban tree species with 94% and 70% accuracy at the tree type and species levels, respectively using high-resolution AVIRIS data.

To estimate CO_2 storage from tree cover requires converting remotely sensed tree crown projection area or diameter into dbh for use in biomass equations. The accuracy of biomass estimates using UGEs is likely to be less than obtained with species-specific biomass equations. Relations between dimensions such as tree crown projection area, crown diameter, dbh, and height have not been well established for urban species. An alternative approach is to determine CO_2 density (CO_2/m^2 tree cover) from ground sampling, perhaps by tree type, and apply these values to classified canopy cover.

3.1.7 Research Goal and Objectives

The goal of this study is to better understand how the choice of approach influences estimates of CO_2 storage in urban forests. Specifically, we compare CO_2 storage estimates obtained with different sets of biomass equations for the same sample of trees. To put our findings in perspective, we compare the variability found in CO_2 stored and sequestered per hectare among estimation approaches for Sacramento's urban forest with the variation found among six other cities.

3.2 Methods

3.2.1 Study Site

The study area consists of the urban areas in the Sacramento metropolitan region. Four counties are included in the region: Sacramento, Yolo, Placer and El Dorado (Fig. 3.1). The experimental unit of analysis involved in this research is the field plot. The total study site area is 131,742 ha.

3.2.2 Field Data

Tree measurement data used in this study were obtained from field measurements following i-Tree Eco protocols and coordinated by the Sacramento Tree Foundation (STF) (Nowak and Crane 2002). In 2007, trained volunteers from STF collected information on 300 random circular plots each 0.04 ha in size (Fig. 3.1). The total number of plots was divided and assigned to teams that had been trained to perform the inventory tasks. Each team sent out letters requesting access to the property when the plot was located on private property. If access was not rejected, the team sampled the plot, obtaining all the parameters described for the UFORE analysis, such as tree species, size (dbh and height), condition, crown light exposure (CLE), position in respect to buildings and land-use.

3.2.3 Allometric Equations

Four sets of allometric equations are described in the following sections.

3.2.3.1 i-Tree Eco

Forest-based biomass equations and the 0.80 multiplier are used to calculate carbon storage and sequestration (Nowak et al. 2002). Hahn's (1984) volumetric formulas are applied to calculate biomass for deciduous trees greater than 94 cm dbh and coniferous trees greater than 122 cm dbh (Nowak et al. 2002).

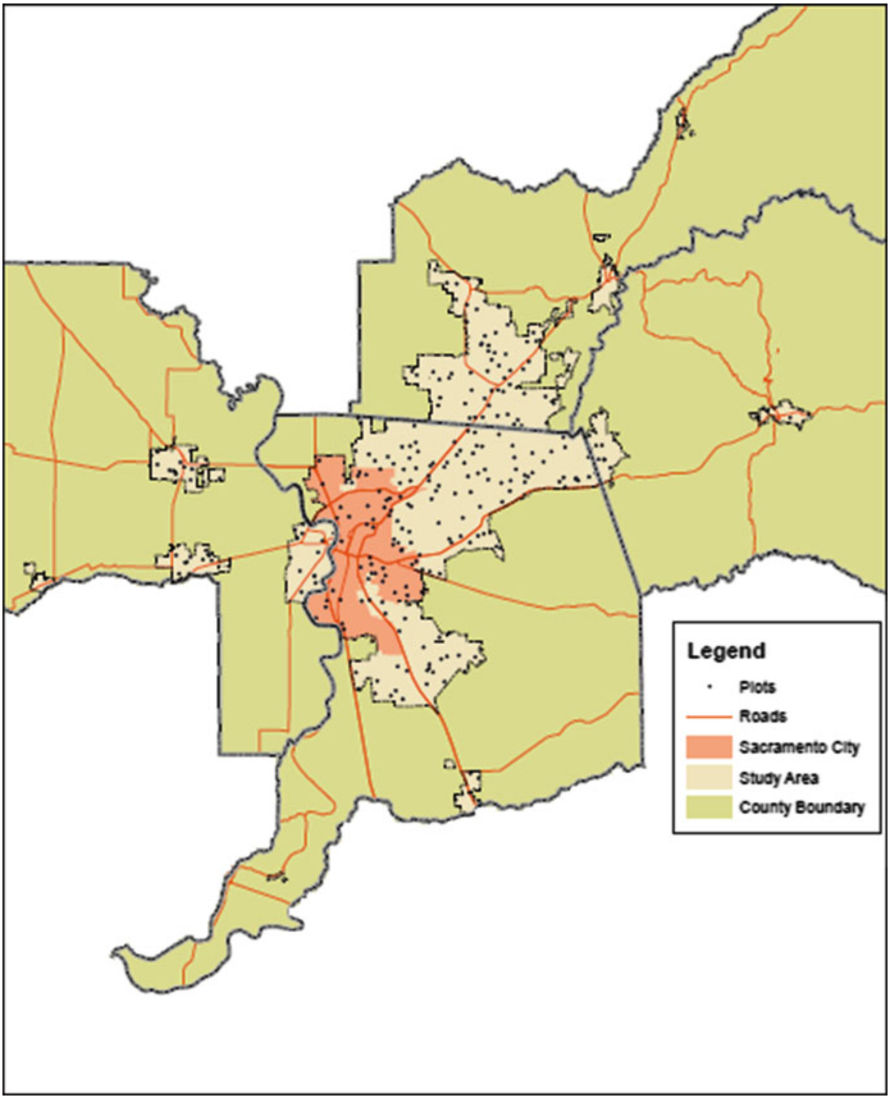

Fig. 3.1 Distribution of UFORE 300 sampling plots in study area (Source: Sacramento study UFORE Draft Report 2010 (Nowak et al. 2010))

Most equations produce dry-weight biomass, some equations compute fresh-weight biomass and are multiplied by species- or genus-specific conversion factors to convert to dry-weight biomass. When a formula is not available for a species, Eco uses the average of results from equations of the same genus. If no genus equations are found, it uses an average of results from all broadleaf or conifer equations.

Eco estimates standardized tree growth based on the number of frost free days and adjusts this base value based on tree condition and location (CLE) to calculate sequestration (Nowak 1994; Nowak et al. 2008). Frost free days are assumed to be 305 for Sacramento, and annual dbh growth ranges from 0.8 to 1.0 cm across all dbh classes. Average height growth is calculated based on formulas from Fleming (1988) and the specific dbh growth factor used for the tree. Growth rates are adjusted based on tree condition as follows: fair to excellent condition – multiplied by 1 (no adjustment), poor condition – 0.76, critical condition – 0.42, dying – 0.15, dead – 0. These growth adjustment factors are based on percent crown dieback and the assumption that less than 25% crown dieback had a limited effect on dbh growth rates (Nowak et al. 2002). Crown light exposure (CLE) provides information on the number of sides of the tree receiving sunlight and ranges from 0 (no full light) to 5 (full light from top and 4 sides).

Gross sequestration is estimated from annual tree growth. Net sequestration incorporates CO_2 emissions due to decomposition after tree death. Emissions are based on the probability of the tree dying within the next year and being removed. Annual removal rates range across dbh classes from 1.4% to 1.9% for condition good to excellent, 3.3% for fair condition, 8.9% for poor condition, 13% for critical, 50% for dying, and 100% for dead (Hoehn 2010).

3.2.3.2 i-Tree Streets

Streets uses the 26 urban-based biomass equations to estimate CO_2 storage for trees in open-grown locations (Pillsbury et al. 1998; McHale et al. 2009). When a formula is not available for a species, Streets uses the closest available urban- or forest-based equation based on taxonomic relationships and wood density characteristics. Forest-based equations are applied with the 0.8 multiplier.

For purposes of comparison, we depart slightly from the Streets protocol by adjusting storage results to account for tree condition. Results from the biomass equations are reduced by 25% for trees in poor or dying condition and 50% for dead trees.

3.2.3.3 CTCC Equations

The CTCC uses biomass equations that are derived almost exclusively from the 26 urban-based equations. Species assignation is different for CTCC because it permits the user to choose one of 16 U.S. climate zones according to the location of the study city. In this study, when a formula is not available for a species from the lists for California's climate zones, a species from the Inland Empire and Central Valley (Climate zones 3 and 4 in the CTCC, Table 3.1) is assigned based on the following criteria: 1-taxonomic, 2-expert opinion on form and growth rate, 3-native or not native. For example, of the 53 species listed for selection in the two climate zones,

Table 3.1 Tree species in Climate zones 3 (Inland Empire) and 4 (Central Valley) of the CUFR Tree Carbon Calculator (McPherson et al. 2008)

Botanic name	Model	Species assigned (Source)
Climate Zone 3-Inland empire		
Brachychiton populneus	= 0.0283168466*(0.00449*(dbhcm/2.54)^2.07041*(3.28*htm)^0.84563)	Magnolia grandiflora (1)
Cinnamomum camphora	= 0.0283168466*(0.00982*(dbhcm/2.54)^2.13480*(3.28*htm)^0.63404)	Cinnamomum camphora (1)
Eucalyptus sideroxylon	= 0.0283168466*(0.00309*(dbhcm/2.54)^2.15182*(3.28*htm)^0.83573)	Eucalyptus globulus (1)
Fraxinus uhdei	= 0.0283168466(0.00129*(dbhcm/2.54)^1.76296*(3.28*htmet)^1.42782)	Fraxinus velutina 'Modesto' (1)
Fraxinus velutina 'Modesto'	= 0.0283168466(0.00129*(dbhcm/2.54)^1.76296*(3.28*htmet)^1.42782)	Fraxinus velutina 'Modesto' (1)
Ginkgo biloba	= 0.0283168466*(0.01177*(dbhcm/2.54)^2.31582*(3.28*htmet)^0.41571)	Liquidambar styraciflua (1)
Jacaranda mimosifolia	= 0.0283168466*(0.011312*(dbhcm/2.54)^2.18578*(3.28*htm)^0.548045)	Jacaranda mimosifolia (1)
Lagerstroemia indica	= 0.0283168466*(0.011312*(dbhcm/2.54)^2.18578*(3.28*htm)^0.548045)	Jacaranda mimosifolia (1)
Liquidambar styraciflua	= 0.0283168466*(0.01177*(dbhcm/2.54)^2.31582*(3.28*htmet)^0.41571)	Liquidambar styraciflua (1)
Liriodendron tulipifera	= 0.0283168466*(0.01177*(dbhcm/2.54)^2.31582*(3.28*htmet)^0.41571)	Liquidambar styraciflua (1)
Magnolia grandiflora	= 0.0283168466*(0.00449*(dbhcm/2.54)^2.07041*(3.28*htm)^0.84563)	Magnolia grandiflora (1)
Phoenix canariensis	= (6*htm + 0.8) + (0.8*htm + 0.9)[a]	Prestoea montana (4)
Phoenix dactylifera	= (6*htm + 0.8) + (0.8*htm + 0.9)[a]	Prestoea montana (4)
Pinus brutia	= 0.0283168466*(0.008573*(dbh/2.54)^2.226808*(3.28*ht)^0.668993)	Pinus radiata (1)
Pinus canariensis	= 0.0283168466*(0.008573*(dbh/2.54)^2.226808*(3.28*ht)^0.668993)	Pinus radiata (1)
Pistacia chinensis	= 0.0283168466(0.00292*(dbhcm/2.54)^2.19157*(3.28*htmet)^0.94367)	Pistacia chinensis (1)
Pinus contorta var. bolanderi	= 0.0283168466*(0.008573*(dbh/2.54)^2.226808*(3.28*ht)^0.668993)	Pinus radiata (1)
Platanus X acerifolia	= 0.0283168466*(0.01043*(dbhcm/2.54)^2.43642*(3.28*htmet)^0.39168)	Platanus X acerifolia (1)
Platanus racemosa	= 0.0283168466*(0.01043*(dbhcm/2.54)^2.43642*(3.28*htmet)^0.39168)	Platanus X acerifolia (1)
Pyrus calleryana	= (EXP(−2.437 + 2.418*(LN(dbhcm))) + EXP(−3.188 + 2.226*(LN(dbhcm))))*0.8[a]	General hardwoods (3)
Quercus agrifolia	= 0.000169*dbhcm^1.956*htmet^0.842	Quercus macrocarpa (2)
Quercus ilex	= 0.0283168466(0.00431*(dbhcm/2.54)^1.82158*(3.28*htmet)^1.06269)	Quecus ilex (1)
Schinus molle	= 0.0283168466(0.00292*(dbhcm/2.54)^2.19157*(3.28*htmet)^0.94367)	Pistacia chinensis (1)
Schinus terebinthifolius	= 0.0283168466(0.00292*(dbhcm/2.54)^2.19157*(3.28*htmet)^0.94367)	Pistacia chinensis (1)
Washingtonia robusta	= (6*htm + 0.8) + (0.8*htm + 0.9)	Prestoea montana (4)

(continued)

Table 3.1 (continued)

Botanic name	Model	Species assigned (Source)
Climate Zone 4- Central valley		
Acer saccharinum	= 0.000238*dbhcm^1.998*htm^0.596	Acer saccharinum (2)
Betula pendula	= a^(b + c*(LOG10(dia^d)))	Betula lenta (5)
Celtis sinensis	= 0.002245*dbhcm^2.118*htm^–0.447	Celtis occidentalis (2)
Cinnamomum camphora	= 0.0283168466*(0.00982*(dbhcm/2.54)^2.13480*(3.28*htm)^0.63404)	Cinnamomum camphora (1)
Fraxinus angustifolia 'Raywood'	= 0.0283168466(0.00129*(dbhcm/2.54)^1.76296*(3.28*htmet)^1.42782)	Fraxinus velutina 'Modesto' (1)
Fraxinus excelsior 'Hessei'	= 0.0283168466(0.00129*(dbhcm/2.54)^1.76296*(3.28*htmet)^1.42782)	Fraxinus velutina 'Modesto' (1)
Fraxinus holotricha	= 0.0283168466(0.00129*(dbhcm/2.54)^1.76296*(3.28*htmet)^1.42782)	Fraxinus velutina 'Modesto' (1)
Fraxinus pennsylvanica 'Marshall'	= 0.000414*dbhcm^1.847*htm^0.646	Fraxinus pennsylvanica (2)
Fraxinus velutina 'Modesto'	= 0.0283168466(0.00129*(dbhcm/2.54)^1.76296*(3.28*htmet)^1.42782)	Fraxinus velutina 'Modesto' (1)
Ginkgo biloba	= 0.0283168466*(0.01177*(dbhcm/2.54)^2.31582*(3.28*htmet)^0.41571)	Liquidambar styraciflua (1)
Gleditsia triacanthos	= 0.000489*dbhcm^2.132*htm^0.142	Gleditsia triacanthos (2)
Koelreuteria paniculata	= 0.0283168466(0.00292*(dbhcm/2.54)^2.19157*(3.28*htmet)^0.94367)	Pistacia chinensis (1)
Lagerstroemia indica	= 0.0283168466*(0.011312*(dbhcm/2.54)^2.18578*(3.28*htm)^0.548045)	Jacaranda mimosifolia (1)
Liquidambar styraciflua	= 0.0283168466*(0.01177*(dbhcm/2.54)^2.31582*(3.28*htmet)^0.41571)	Liquidambar styraciflua (1)
Magnolia grandiflora	= 0.0283168466*(0.00449*(dbhcm/2.54)^2.07041*(3.28*htm)^0.84563)	Magnolia grandiflora (1)
Phoenix canariensis	= (6*htm + 0.8) + (0.8*htm + 0.9)[a]	Prestoea montana (4)
Phoenix dactylifera	= (6*htm + 0.8) + (0.8*htm + 0.9)[a]	Prestoea montana (4)
Pinus brutia	= 0.0283168466*(0.008573*(dbh/2.54)^2.226808*(3.28*ht)^0.668993)	Pinus radiata (1)
Pistacia chinensis	= 0.0283168466(0.00292*(dbhcm/2.54)^2.19157*(3.28*htmet)^0.94367)	Pistacia chinensis (1)
Pinus contorta var. bolanderi	= 0.0283168466*(0.008573*(dbh/2.54)^2.226808*(3.28*ht)^0.668993)	Pinus radiata (1)
Pinus radiata	= 0.0283168466*(0.005325*(dbh/2.54)^2.226808*(3.28*ht)^0.668993)	Pinus radiata (1)
Pinus thunbergiana	= 0.0283168466*(0.005325*(dbh/2.54)^2.226808*(3.28*ht)^0.668993)	Pinus radiata (1)

Platanus hybrida	= 0.0283168466*(0.01043*(dbhcm/2.54)^2.43642*(3.28*htmet)^0.39168)	Platanus hybrida (1)
Pyrus calleryana 'Bradford'	= (EXP(−2.437 + 2.418*(LN(dbhcm))) + EXP(−3.188 + 2.226*(LN(dbhcm))))*0.8[a]	General hardwoods (3)
Pyrus kawakamii	= (EXP(−2.437 + 2.418*(LN(dbhcm))) + EXP(−3.188 + 2.226*(LN(dbhcm))))*0.8[a]	General hardwoods (3)
Quercus ilex	= 0.0283168466(0.00431*(dbhcm/2.54)^1.82158*(3.28*htmet)^1.06269)	Quercus ilex (1)
Washingtonia robusta	= (6*htm + 0.8) + (0.8*htm + 0.9)	Prestoea montana (4)
Zelkova serrata	= 0.0283168466(0.00666*(dbhcm/2.54)^2.36318*(3.28*htmet)^0.55190)	Zelkova serrata (1)

Source: (1) Pillsbury et al. (1998), (2) McHale et al. (2009), (3) Harris et al. (1973), (4) Frangi and Lugo (1985), (5) Jenkins et al. (2003a, b)/Martin

[a]Equation predicts dry weight instead of volume

biomass for two pears (*Pyrus calleryana* and *kawakami*) and three palms (*Phoenix canariensisis* and *dactylifera*, *Washingtonia robusta*) is calculated with forest-based equations (general hardwoods and palms equations). Carbon dioxide storage is not adjusted for tree condition or with the 0.8 multiplier. Total dry weight biomass is calculated and converted to CO_2.

Carbon dioxide sequestration is calculated using growth curves developed from intensive measurements on a sample of about 1,000 street trees representing the 20 predominant species measured in each of the California reference cities. Sequestration is not adjusted for condition or mortality.

3.2.3.4 Urban General Equations (UGEs)

A set of UGEs equations was developed to compare with results from species-specific equation sets. Trees are classified into three types that are readily distinguished with remote sensing: broadleaves, conifers and palms. Tree volume equations are derived exclusively from the 26 urban-based formulas and converted to biomass equations (Table 3.2). Total dry weight biomass is converted to CO_2 storage. There is no tree condition adjustment and no species assignation is applied.

Development of Urban General Biomass Equations

Both sets of urban equations were converted to the International System of Units (SI units). Pillsbury's 15 equations were corrected for standard error. The publication included antilogarithmic error that had to be converted to root mean square error (RMSE). McHale's equations were corrected as well, although the publication included RMSE values for each equation. The RMSE values of the 26 equations were used in calculating new coefficients that accounted for the error. Logarithmic expressions of each of the 26 new coefficients were taken. Coefficients b and c (the later only existing in the sets of equations with height) were left unchanged.

The logarithmic expressions of all new coefficients *"a"* were sorted by tree type and the maximum and minimum values plotted to observe differences between tree types. The same procedure was repeated for coefficients b and c. The separation between broadleaf evergreen species and broadleaf deciduous species was not clear, so both groups were combined into a single tree type called broadleaf.

Differences among coefficients were evident for species belonging to broadleaf and conifer tree types. However, only two of the published equations were for conifers. More data on conifers is necessary to better identify and explain causes for coefficient differences.

The final urban general equations (UGEs) have the same format as the species-specific equations [$\text{biomass} = A^*(\text{dbh})^B$] and [$\text{biomass} = A^*(\text{dbh})^{B*}(\text{height})^C$]. Coefficient *A, B and C* were calculated by averaging the logarithmic expressions of the new *"a"* "*b"* and "*c"* coefficients.

Table 3.2 26 Available urban allometric equations from California-Colorado

Tree species	Volume equation=a*(dbh)^b	R^2	RMSE	Source	dbh min (cm)	dbh max (cm)	height min (m)	height max (m)
Fraxinus pennsylvanica	0.00059*(dbh)^2.206	0.987	0.18	McHale	1	140	Not reported	Not reported
Gleditsia triancanthos	0.00051*(dbh)^2.22	0.988	0.19	McHale	1	140	Not reported	Not reported
Tilia cordata	0.00094*(dbh)^2.042	0.953	0.26	McHale	1	140	Not reported	Not reported
Quercus macrocarpa	0.00024*(dbh)^2.425	0.938	0.37	McHale	1	140	Not reported	Not reported
Celtis occidentalis	0.0014*(dbh)^1.928	0.959	0.29	McHale	1	140	Not reported	Not reported
Ulmus americana	0.0018*(dbh)^1.869	0.924	0.27	McHale	1	140	Not reported	Not reported
Acer platanoides	0.0019*(dbh)^1.785	0.94	0.28	McHale	1	140	Not reported	Not reported
Ulmus pumila	0.0049*(dbh)^1.613	0.874	0.46	McHale	1	140	Not reported	Not reported
Populus sargentii	0.0021*(dbh)^1.873	0.991	0.18	McHale	1	140	Not reported	Not reported
Gymnocladus dioicus	0.00042*(dbh)^2.059	0.816	0.41	McHale	1	140	Not reported	Not reported
Acer saccharinum	0.00036*(dbh)^2.292	0.964	0.33	McHale	1	140	Not reported	Not reported
Acacia longifolia	0.048490*(dbh)^2.347250	0.938	0.22	Pillsbury	15	57.2	9.3	16.3
Liquidambar styraciflua	0.030684*(dbh)^2.560469	0.979	0.14	Pillsbury	14	54.4	7.3	20
Eucalyptus globulus	0.055113*(dbh)^2.436970	0.968	0.24	Pillsbury	15.5	130	14.1	43.9
Cinnamomum camphora	0.031449*(dbh)^2.534660	0.97	0.16	Pillsbury	13.2	68.8	5.2	17.1
Ceratonia siliqua	0.066256*(dbh)^2.128861	0.91	0.25	Pillsbury	15.5	71.4	4.7	10.8
Ulmus parvifolia chinensis	0.028530*(dbh)^2.639347	0.903	0.2	Pillsbury	17.3	55.9	7.6	18.9
Pistacia chinensis	0.019003*(dbh)^2.808625	0.958	0.22	Pillsbury	12.7	51.3	6.7	15.8
Quercus ilex	0.025169*(dbh)^2.607285	0.938	0.22	Pillsbury	12.7	52.1	5.2	17.1
Jacaranda mimosifolia	0.036147*(dbh)^2.486248	0.949	0.17	Pillsbury	17.3	59.7	6.9	17.5
Platanus acerifolia	0.025170*(dbh)^2.673578	0.965	0.2	Pillsbury	15.5	73.9	7.9	27.9
Fraxinus velutina 'Modesto'	0.022227*(dbh)^2.633462	0.94	0.25	Pillsbury	14.5	84.8	5.6	22.6
Cupressus macrocarpa	0.035598*(dbh)^2.495263	0.98	0.21	Pillsbury	15.7	146.6	8.1	30.8
Pinus radiate	0.019874*(dbh)^2.666079	0.969	0.24	Pillsbury	16.8	105.4	5.5	32.2
Zelkova serrata	0.021472*(dbh)^2.674757	0.969	0.19	Pillsbury	14.5	86.4	6.1	21
Magnolia grandiflora	0.022744*(dbh)^2.622015	0.958	0.22	Pillsbury	14.5	74.2	5.8	18.9

Source of equations: Pillsbury et al. (1998), McHale et al. (2009)

For the purpose of this study, dbh based equations were used for comparisons because dbh can be derived from tree crown projection area obtained from remotely sensed imagery. More research is needed to identify relations between these dimensions.

The biomass equation for palms was an equation for *Prestoea montana* based on destructive measurements for individuals of this species growing in a Puerto Rican floodplain forest (Frangi and Lugo 1985). Biomass equations for palms growing in U.S. cities are not available. Because palms trees do not have secondary growth the only parameter is tree height. The UGEs that will be used subsequently are:

$$\text{Broad leaf}_{\text{biomass}}(\text{only dbh}) = 0.16155 * (\text{dbh})^\wedge 2.310647 \tag{3.3}$$

$$\text{Conifer}_{\text{biomass}}(\text{only dbh}) = 0.035702 * (\text{dbh})\ ^\wedge\ 2.580671 \tag{3.4}$$

$$\text{Palm}_{\text{biomass}} = 1.282 * (7.7 * \text{ht} + 4.5) \tag{3.5}$$

These equations estimate total dry weight (kg, above and below ground) based on measured dbh (cm) and tree height (m) for palms.

3.2.4 Scale-Up

An area-based approach is used to scale-up CO_2 storage estimates from the 300 plots to the entire study area. Total storage for the 300 plots (12.1 ha) are proportionally scaled up to the entire study area (131,742 ha) using the scalar 10,851 (131,742/12.1). The total CO_2 storage density (kg/ha) for all plots is multiplied by the same scalar as well. One exception is the Eco model, which includes tree density as well as area in the scale-up calculation (Nowak 1994; Hoehn 2010).

3.3 Results

3.3.1 Comparison of UGEs with Other Biomass Equations

UGEs developed with urban-based biomass equations are compared with general forest biomass equations for hardwoods and softwoods using the same dbh or dbh-height data. The results, plotted in Fig. 3.2, reveal that at sizes larger than 35 cm dbh UGE predicted above-ground biomass is about 25% less than predicted with forest-based general equations for hardwoods and about 10% less for than for softwoods. Differences are less noticeable for smaller sized trees.

Conifers accumulate less biomass than broadleaves through their growth cycle, due in part to lower wood density. There is a small difference in biomass storage estimates between urban broadleaf tree types. However, urban broadleaf evergreens store a little less biomass than urban broadleaf deciduous trees.

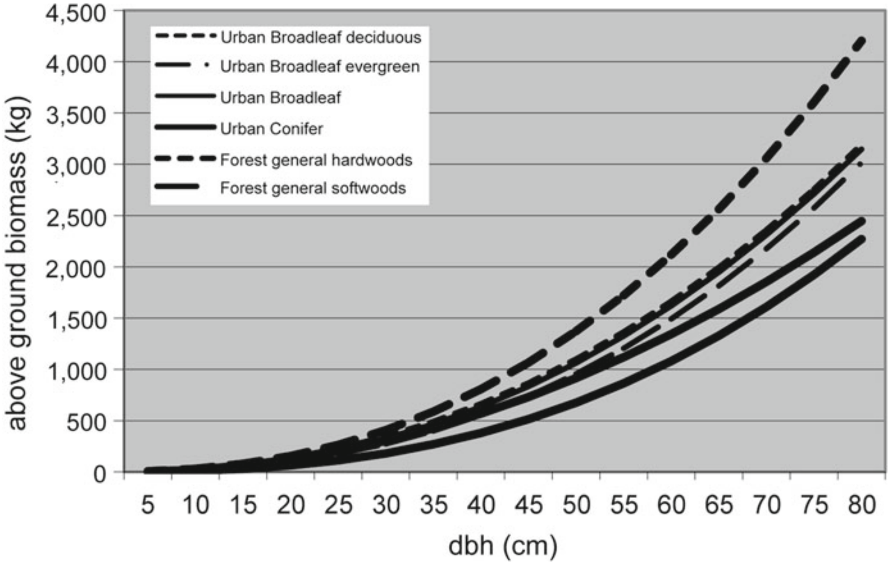

Fig. 3.2 Urban- and forest-based general equations

Table 3.3 Carbon dioxide storage (t), sequestration (t), and density (t/ha) for all plots (12.1 ha) and the study site (131,742 ha, area-based scale up)

Biomass equations	Plot storage	Plot sequestration	Study area storage	Study area sequestration	Density storage	Density sequestration
Eco	458.1	22.0	4,989,515	238,589	38.2	1.8
Streets	591.0		6,412,544		48.7	
CTCC	589.9	34.1	6,400,723	370,413	48.6	2.8
UGE	469.8		5,098,100		38.7	

3.3.2 Plot Level: Comparison of Storage Estimations

The comparison of CO_2 storage and sequestration calculations for the 640 trees in the 300 plots is presented in Table 3.3. i-Tree Streets (591 t) and CTCC (590 t) storage estimates are very similar and 26% greater than the UGE value (470 t). The i-Tree Eco CO_2 storage estimate (458 t) is about 3% less than the UGE value. The highest estimates for Streets and CTCC are 29% greater than the lowest estimate for Eco. Carbon dioxide storage density values range from 38 to 49 t/ha (Table 3.2).

Sequestration estimates are only available for the Eco and CTCC equation sets because they have associated tree growth data. The CTCC estimate (34.1 t) is 55% higher than the Eco value (22.0 t). Carbon dioxide sequestration density values range from 1.8 to 2.8 t/ha.

3.3.3 Scale Up Results

Carbon dioxide storage and sequestration differences noted at the plot level are also reflected at the regional or study area level (Table 3.2). Storage values obtained with Streets (6.41 Mt) and CTCC (6.4 Mt) equation sets are similar and substantially greater than the Eco estimate (4.9 Mt). The estimate obtained with the UGE (5.1 Mt) is about 3% greater than the Eco estimate.

3.4 Discussion

This study found a maximum 29% difference in plot-level CO_2 storage among the four sets of biomass equations. As expected, i-Tree Eco equations produced the lowest estimate (458 t), presumably because forest-based equations are used exclusively with application of the 0.8 multiplier to open-grown trees. The UGEs produced an intermediate estimate (470 t), and the CTCC and Streets equations produced substantially larger estimates that were very similar (590 and 591 t).

3.4.1 Differences by Species

To explain causes for different estimates it is useful to examine differences among species that are most important by virtue of their relative abundance and size. For example, Interior live oak (*Quercus wislizenii*) is not the most abundant species, but it stores the most CO_2 according to all four sets of equations (Fig. 3.3). Estimated CO_2 storage for the species ranged from 82 t (UGEs) to 142 t (CTCC).

According to three sets of equations (Eco, Streets, CTCC), the next most important species, Blue oak (*Quercus douglasii*) stores nearly one-half as much CO_2 as Interior live oak, but the UGE shows a small difference. Estimates from the UGE's tended to be among the lowest for the oaks, but among the highest for other important species such as Alder (*Alnus spp.*), White mulberry (*Morus alba*), London planetree (*Platanus acerifolia*) and Atlas cedar (*Cedrus deodara*). Similarly, storage estimates from the Eco equations were the lowest for oaks, but among the highest for London planetree, olive (*Olea europaea*), and several other species.

3.4.2 Effects of Different Biomass Equations

A more detailed picture of the variability among estimates of stored and sequestered CO_2 is presented in Tables 3.4 and 3.5. The values are calculated by species using

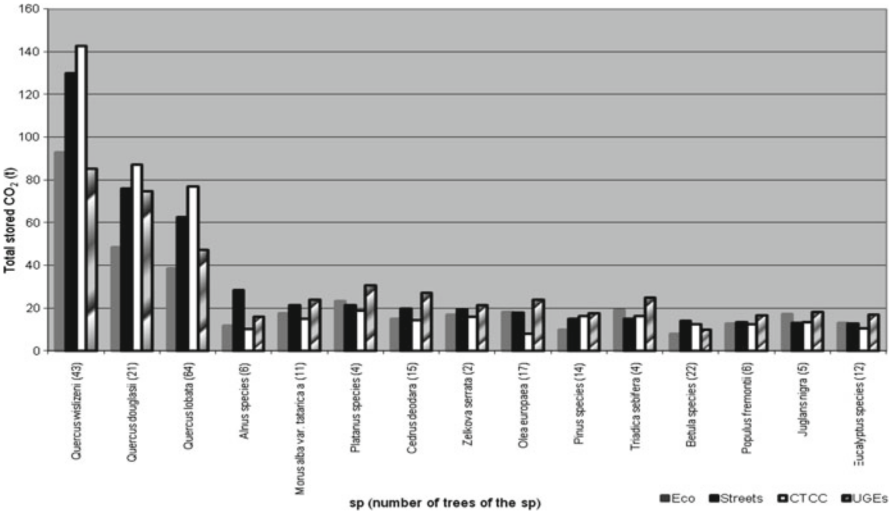

Fig. 3.3 Carbon dioxide storage estimates by species calculated with four sets of biomass equations for sampled trees (number in brackets)

the species assignments listed in Table 3.5 and the mean dbh and height for all trees sampled. They do not account for adjustments based on CLE, condition, or mortality. The maximum difference is expressed as a percentage: (High value/Low value) × 100, where 100 is no difference. The minimum difference is the difference between the two closest values.

For CO_2 storage, the minimum difference is less than 5% for five species, but greater than 10% for the remaining five species. The maximum difference is at least two-fold for all species except olive, and exceeds three-fold for five species. An eight-fold maximum difference exists for alder between the Streets (4,320 kg) and Eco (539 kg) estimates. Both minimum and maximum differences are relatively high for the three oak species, who together account for about one-half of all CO_2 stored. There are no discernable trends in terms of a set of equations always producing estimates that are the highest or lowest across all species.

In the comparison of sequestration rates among species (Table 3.4), differences exceed ten-fold for two species and are five- to six-fold for three other species. Differences are relatively high for the oaks, alder and pine (*Pinus spp*.). Here there is a clear trend, with CTCC estimates always greater than Eco estimates. Eco values are for gross sequestration, so the differences in Table 3.4 cannot be due to reductions for tree condition and projected mortality in Eco. More likely explanations are differences in tree growth rates and selection of biomass equations.

Results in Table 3.4 illustrate how selection of biomass equations influence storage estimates. Using the same tree dbh and height but different biomass equation for the same species can result in dramatically different estimates.

Table 3.4 Carbon dioxide stored and sequestered, and differences among the four sets of equations calculated on a per tree basis with mean dbh and height for the most important species

			Stored (kg)						Sequestered (kg/year)		
Common name	dbh (cm)	Height (m)	Eco	Streets	CTCC	UGE	Min diff (%)	Max diff (%)	Eco	CTCC	Diff (%)
Interior live oak	29.9	11.3	589	991	1,531	762	129	260	23.5	136.6	582
Blue oak	34.2	10.3	627	1,162	2,118	1,039	112	338	15.7	163.5	1,045
California white oak	19.3	10.5	163	395	535	277	169	327	10.8	74.4	687
Alder	44.2	15.7	539	4,320	1,454	1,879	129	801	21.5	111.1	517
White mulberry	35.6	10.8	431	1,118	680	1,140	102	265	16.6	47.3	284
London planetree	23.8	11.8	135	312	302	449	103	334	11.2	49.0	440
Atlas cedar	8.1	3.7	6	8	11	14	129	241	1.4	5.0	374
Japanese zelkova	84.1	21.1	2,280	9,705	8,114	8,307	102	426	74.1	105.3	142
Olive	24.9	8.1	288	268	265	499	101	188	12.7	33.8	267
Pine	29.7	10.7	190	401	463	414	103	244	6.6	77.7	1,171

Table 3.5 Species assignation differences among the four sets of equations

Species name	i-Tree Eco	i-Tree Streets	CTCC	UGEs
Quercus wislizeni	Q. wislizeni (Pillsbury and Kirkley 1984)	Quecus ilex (Pillsbury et al. 1998)	Quercus macrocarpa (McHale et al. 2009)	Broadleaves UGE (Eq. 3.3)
Q douglasii	Q. douglasii (Pillsbury and Kirkley 1984)	Quecus ilex (Pillsbury et al. 1998)	Quercus macrocarpa (McHale et al. 2009)	Broadleaves UGE (Eq. 3.3)
Q. lobata	Average oaks (Nowak and Crane 2002)	Quecus ilex (Pillsbury et al. 1998)	Quercus macrocarpa (McHale et al. 2009)	Broadleaves UGE (Eq. 3.3)
Alnus sp	Average broadleaves (Nowak and Crane 2002)	Populus sargentii (McHale et al. 2009)	Eucalyptus sideroxylon (Pillsbury et al. 1998)	Broadleaves UGE (Eq. 3.3)
Morus alba	Average broadleaves (Nowak and Crane 2002)	Fraxinus velutina 'Modesto' (Pillsbury et al. 1998)	Fraxinus velutina 'Modesto' (Pillsbury et al. 1998)	Broadleaves UGE (Eq. 3.3)
Platanus sp	Average broadleaves (Nowak and Crane 2002)	Platanus hybrida (Pillsbury et al. 1998)	Platanus hybrida (Pillsbury et al. 1998)	Broadleaves UGE (Eq. 3.3)
Cedrus deodara	Average cedars (Nowak and Crane 2002)	Pinus radiata (Pillsbury et al. 1998)	Pinus radiata (Pillsbury et al. 1998)	Conifers UGE (Eq. 3.4)
Zelkova serrata	Average broadleaves (Nowak and Crane 2002)	Zelkova serrata (Pillsbury et al. 1998)	Zelkova serrata (Pillsbury et al. 1998)	Broadleaves UGE (Eq. 3.3)
Olea europaea	Average broadleaves (Nowak and Crane 2002)	Fraxinus velutina 'Modesto' (Pillsbury et al. 1998)	Fraxinus velutina 'Modesto' (Pillsbury et al. 1998)	Broadleaves UGE (Eq. 3.3)
Pinus sp.	Average pines (Nowak and Crane 2002)	Pinus radiata (Pillsbury et al. 1998)	Pinus canariensis (Pillsbury et al. 1998)	Conifers UGE (Eq. 3.4)

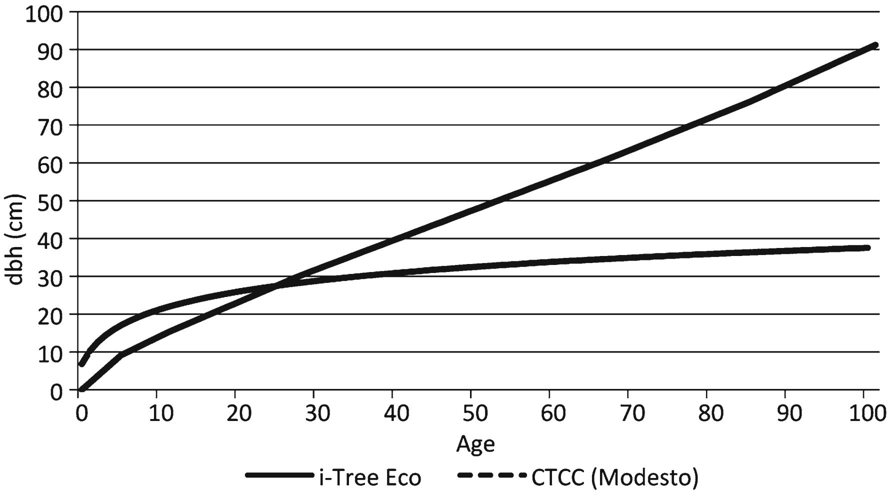

Fig. 3.4 Eco and CTCC growth curves for Green ash

3.4.3 Single Species Example

Carbon dioxide sequestration estimates produced by Eco and CTCC are influenced by tree growth and size, as well as selection of the allometric equation. The extent to which these factors influence sequestration is shown for the same species, Green ash (*Fraxinus pennsylvanica*) using unadjusted data from the i-Tree Eco and CTCC models (Fig. 3.4). The Eco growth curve shows initial rapid growth for 5 years followed by moderate growth that increases linearly until year 100. The CTCC growth model starts with a larger tree, but the growth becomes quite slow after 10 years. After 25 years the size of the tree modeled in Eco surpasses the CTCC tree, and after 80 years is twice the dbh of the CTCC tree.

The amount of CO_2 stored by the same size trees using the different biomass equations applied in Eco and CTCC shows a similar trend (Fig. 3.5). Carbon dioxide stored by the CTCC tree is greater than the Eco tree initially, but becomes less once the tree reaches 40 cm dbh and its growth ceases. In Eco, the sequestration rate gradually increases with tree size. After the tree surpasses 55 cm dbh, it begins to store more CO_2 than estimated by the CTCC tree.

The Eco growth model uses a base growth increment (0.83 cm/year) that is adjusted based on frost free days, CLE and condition. As explained by Nowak (1994) growth is also adjusted based on dbh. Growth rates are grouped by genera and dbh. Averages are used as base growth rates for specific land uses and are then altered based on length of growing season. The base tree growth rate comes from trees measured in northern latitudes, and may well underestimate growth in California. Growth rates used by CTCC are based on data measured for street and park trees in California cities.

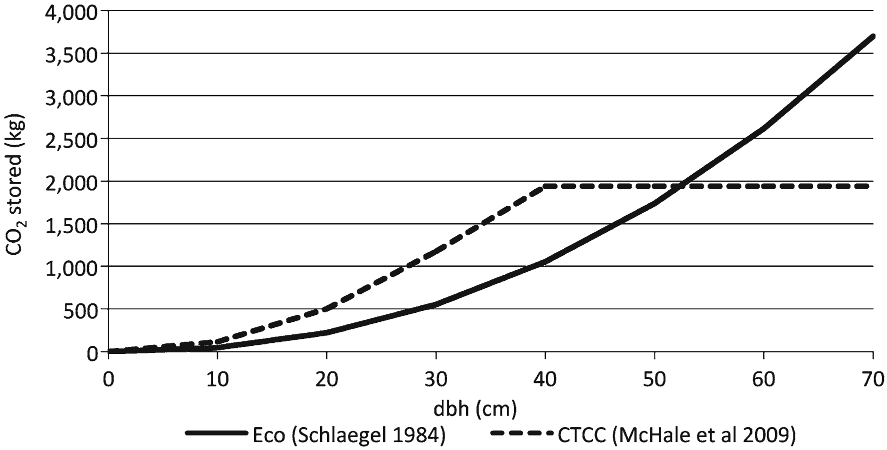

Fig. 3.5 CO_2 storage by dbh in Green ash

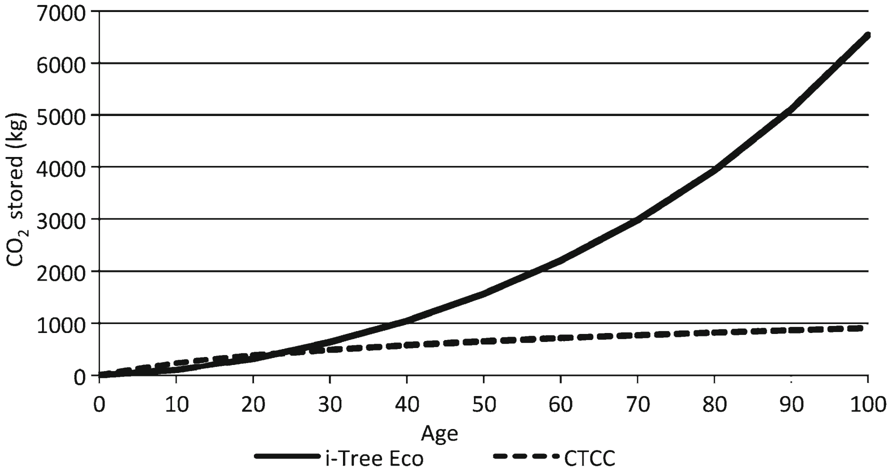

Fig. 3.6 CO_2 storage by age in Green ash

The amount of CO_2 stored as a function of tree age incorporates effects of tree growth and size with solutions produced by each allometric equation (Fig. 3.6). Differences between Eco and CTCC are small for the first 30 years, but become pronounced with tree age. At 100 years the green ash modeled with Eco has stored over six times the amount of CO_2 as the ash modeled with the CTCC.

Because the UGEs produced relatively low CO_2 storage estimates for the most important species, it is not surprising that they produced a relatively low estimate for all 640 sampled trees. UGE storage estimates could be relatively higher compared to the other approaches if the tree population had a different distribution of

Table 3.6 Tree density, stored and sequestered carbon dioxide per hectare for several cities

City	Trees/ha	Storage CO_2 t/ha	Seq. CO_2 t/ha/year	Reference
Sacramento, USA[a]	68	91.9	2.8	McPherson (1998)
Atlanta, USA	276	131.2	4.5	Nowak and Crane (2002)
New York, USA	65	56.3	1.8	Nowak and Crane (2002)
Chicago, USA[b]	69	52.0	2.4	Nowak (1994)
Miami-Dade, USA	288	43.1	3.2	Escobedo et al. (2010)
Gainesville, USA	528	117.1	4.5	Escobedo et al. (2010)
Chuncheon, Korea[b]	150	4.7	0.6	Jo (2002)

[a]City and suburban sectors only
[b]City only

importance among species and different biomass equation species assignments. It appears that the accuracy of UGE estimates relative to estimates derived from species-based equations depends on the population structure and idiosyncrasies of species and biomass equation assignments.

3.4.4 Differences Among Cities

The CO_2 storage and sequestration results from this study are difficult to compare with other studies because of differences in forest composition, age structure, and scope of the analyses. Forests with low tree density and abundant softwoods will store less CO_2 than high density, hardwood forests. Population density and the extent of urbanization influence urban forest density. Forests in old parts of the city often store more CO_2 than forests in new development because trees are mature. However, sequestration rates may be greater in younger areas where trees are growing rapidly. The scope of the study influences results because it may include CO_2 emissions from anticipated mortality and tree care activities. Some studies include reduced emissions from energy savings. Also, some studies include storage from interface forests and peri-urban natural areas, while others are limited to developed areas. To facilitate comparisons across cities mean CO_2 storage and sequestration rates are presented per hectare (Table 3.6).

Compared to the previous Sacramento study (McPherson 1998), the study area for this analysis is much larger, and includes a larger amount of undeveloped land in agricultural and other non-forest uses. Sampling intensity was greater in the previous study, with 460 plots in 61,000 ha versus 300 plots in 131,000 ha. In comparison with the previous Sacramento study (Table 3.6), the scaled-up data from this study found relatively low tree density (53/ha), CO_2 storage (38–49 t/ha) and sequestration (1.8–2.8 t/ha) rates. The mean dbh measured in the current study is 18 cm, or about one-half the size recorded in the previous Sacramento study (39 cm). Thus, the lower amount of CO_2 storage estimated in the current study may be partially explained by lower tree density and younger, smaller trees on average. To some extent, this result may be an artifact of differences in the area sampled and sampling intensity.

Sacramento tree density, stored and sequestered CO_2 rates are at the low end compared with the temperate climate cities of New York and Chicago. The range of variability reported here for different sets of equations does not exceed the ranges encompassed by the cities in Table 3.6. In this study, Sacramento's estimated CO_2 storage ranged from 38 to 49 t/ha, while it ranged from 4.7 to 131.2 t/ha for the six other cities cited (Table 3.6). Sacramento urban forest's estimated annual sequestration rate ranged from 1.8 to 2.8 t/ha, compared to 0.6 to 4.5 t/ha for the cities. Cities in the southeast USA have higher tree densities and sequestration rates. Storage rates are higher for Atlanta and Gainesville, but less for Miami-Dade, where stands of invasive punktree (*Melaleuca quinquenervia*) are the largest CO_2 sink. In contrast, storage and sequestration rates are low in Chuncheon, Korea, although tree density is relatively high compared to temperate climate cities in the USA.

3.5 Conclusion

This study found substantial variability among four approaches for calculating the amount of CO_2 stored and sequestered by Sacramento's urban forest. Storage estimates differed by a maximum of 29% and ranged from 38 to 49 t/ha. The two sequestration estimates differed by 55%, ranging from 1.8 to 2.8 t/ha. Although error associated with these storage estimates is considerable, its importance is diminished when one considers other sources of error, such as sampling, measurement, growth modeling, and biomass equation selection.

The variability associated with these four approaches is not great when compared to the variability in CO_2 storage and sequestration densities among cities. The maximum differences in CO_2 storage and sequestration rate differences among approaches are 11 and 1 t/h, respectively. These are relatively small amounts compared to the maximum differences reported for six other cities of 127 and 3.9 t/ha, respectively (Table 3.6). Differences among cities reflect differences in forest composition, age structure, and scope of the analyses, as well as differences in biomass equations, tree growth modeling, sampling, and measurement.

Explanations for differences observed among approaches are difficult to determine, although some trends are apparent. Eco produced the lowest storage estimate, perhaps because it relied exclusively on forest-based equations and applied a 0.80 correction factor to open-grown trees. The storage estimates produced by Streets and CTCC were the highest, perhaps reflecting ubiquitous application of urban-based biomass equations. The UGEs produced relatively low estimates of CO_2 storage. This result may be idiosyncratic to this sample of 640 trees because UGE estimates are more sensitive to the population's species composition and structure than do estimates derived from species-based equations.

Eco produced lower estimates of CO_2 sequestration rates than the CTCC across a range of species. Reductions for tree condition and projected mortality may partially explain the difference.

Also, selection of biomass equations to apply for each species was found to substantially influence storage estimates using the same input dimensions but different equations for the top ten species.

An examination of the roles of tree growth modeling and biomass equation selection for a green ash tree illustrated their importance. The Eco tree stored more CO_2 after 30 years than the CTCC tree, largely due to increased growth projected over the 100 year period. The analysis illustrated the how the dynamic interaction between tree growth and biomass storage rate can influence the temporal stream of sequestration in complex ways.

Based on these results we conclude that applying UGEs to remotely sensed data that accurately classify broadleaf, conifer and palm tree types in the Sacramento region is likely to produce conservative results compared to results from urban-based species-specific equations. The robustness of this result needs to be tested with different tree populations because of the large variability associated with assigning a limited number of urban-based biomass equations to diverse assemblages of species. This result suggests that there is promise of obtaining initial estimates of carbon dioxide storage by urban forests using UGEs for tree types identified with remote sensing when resources do not allow for field sampling. Further research is needed to establish relations between remotely-sensed tree crown projection area and dbh values required for biomass calculation. Of course, ground-based inventories remain necessary for more accurate estimates of CO_2 storage and for municipal forest management and health monitoring purposes.

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Chapter 4
Terrestrial Carbon Management in Urban Ecosystems and Water Quality

Klaus Lorenz and Rattan Lal

Abstract The hydrology of urban ecosystems is drastically different from those of natural, rural ecosystems. Urbanization deliberately alters natural hydrosystems for domestic uses, industrial processes, sanitation, and protection from floods, hurricanes, and tsunamis because most ancient urban centers were sited along rivers and deltas. The amount of water extracted in the natural environment for human use has tripled over the last 50 years. This water withdrawal is likely to increase in the future as the projected increase in global energy production by 60% until 2030 requires more water. In modern urban ecosystems, water quality is also affected by increased loadings of nutrients, metals, pesticides, and other contaminants in urban runoff, municipal and industrial discharges. Drastic alteration in hydrological fluxes is caused by the increase in impervious surface cover which decreases infiltration, increases surface runoff, and transports pollutants into streams. In 2000, for example, the impervious surface cover of the conterminous U.S. was 83,749 km^2, and is projected to increase to 114,070 km^2 by 2030, approaching the size of Ohio. Thus, reducing the impervious surface cover can moderate the extreme hydrologic conditions observed in urban centers. Managing urban storm water and establishing green roofs are other measures to improve urban hydrology. Stormwater runoff reduction by green roofs, for example, can be as much as 27% and 87% of annual precipitation. By establishing closed nutrient and water cycles in urban centers, the quality of water efflux from urban ecosystems can be improved. Increasing the levels of soil organic matter (SOM), in particular, drives

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microbial activity and nitrate (NO_3^-) removal, and retains nutrients and contaminants in the soil. This can be achieved by increasing the urban area covered by vegetation and soil, increasing pervious sealing cover types, establishing green roofs on buildings, restoring degraded urban soils and enhancing the SOM sink.

Keywords Urban hydrology • Urban soils • Impervious surface • Soil organic matter • Green roofs

4.1 Introduction

Human activities have exacerbated species extinctions, invasions, and introductions and domestications. Anthropogenic activities have also accelerated soil erosion, altered fire frequency and hydrology, and caused profound changes in net primary productivity (NPP) and other key biogeochemical and ecosystem processes (Ellis et al. 2010). Urbanization (i.e., the expansion of urban land uses, including commercial, industrial, and residential uses) is one of the most dramatic and dynamic global human alteration of ecosystems (Grimm et al. 2008). About 60% of the city population or 29% of the global population resides in anthropogenic biomes (anthromes) (Ellis et al. 2010). For the first time in history, more than 50% of the global population lives in urban centers, towns and settlements (UNFPA 2007). The urban population is projected to increase rapidly with attendant increases in demands on nearby and distinct ecosystems. Specifically, urban lands comprise the most intensively transformed lands on earth, and urban land cover changed from 0.01% of the global ice-free land area in 1700 to 0.5% in 2002 (Ellis et al. 2010; Schneider et al. 2009). Globally, urbanization is now the primary process of land cover transformation (Pavao-Zuckerman and Byrne 2009). For example, it is estimated that an additional land area of the size of California will be converted globally to urban ecosystems by 2030 (Angel et al. 2005). Furthermore, urban sprawl (i.e., urban encroachment of rural land) directly affects a quarter of the territory of the European Union (European Environment Agency 2006). Most importantly, urbanization takes place on Earth's most fertile lands (Seto and Sheperd 2009). Thus, urban ecosystems require agricultural production in other areas, and also resources such as water, energy and transportation infrastructure. However, the scientific understanding of coupled anthropogenic and ecologic processes is scanty (Ellis et al. 2010).

Since the 1960s, the number of people under water shortage has increased rapidly (Kummu et al. 2010). Thus, the effects of changes in global population on water shortage are roughly four times more important than changes in water availability as a result of long-term climatic change. Together with the rapid urban population growth the amount of water extracted in the natural environment for human use (i.e., water withdrawal) has tripled over the last 50 years (World Water Assessment Programme 2009). Thus, urbanization creates some of the drastic pressures on water resources quantity and quality. Specifically, the hydrosystem is modified as water resources in cities have to be managed for energy and food production, transportation, flood control, drinking water, sanitation, and commercial and industrial activities.

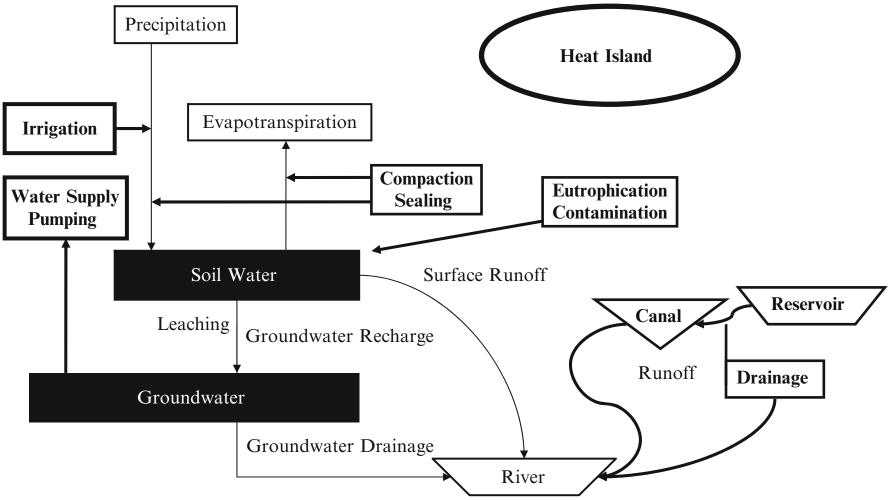

Fig. 4.1 Water flux in urban ecosystems and main effects of urbanization (Modified from L'Vovich and White 1990)

In 2000, industrial use accounted for 20% and domestic (urban) use for about 10% of total water withdrawal, respectively (World Water Assessment Programme 2009). However, pressures on urban water resources are likely to increase due to rapid urbanization, population density increases, economic growth, technological changes and changing consumption patterns. For example, 60% more energy is needed globally in 2030 compared to 2002, and this drastic increase in energy production will require more water. In addition, climate change may increase extreme events causing more floods and droughts in regions also characterized by rapid urbanization. For example, climate change is projected to further aggravate floods and droughts in Mexico City resulting from land use changes associated with urban growth (Lankao 2010).

Local and regional hydrosystems are also affected by urbanization (Grimm et al. 2008). Specifically, water quantity and quality are modified by urban land use and cover change as many ancient urban centers had been sited along rivers and deltas. The designed or altered streams, rivers, flood channels, canals and other hydrosystems neither replicate the aquatic ecosystems they replace nor preserve the ecosystem services lost (Fig. 4.1). The consistently observed ecological degradation of streams draining urban land has been described as 'urban stream syndrome' (Meyer et al. 2005). Indications of the urban stream syndrome include: (i) a flashier hydrograph (i.e., more frequent, larger flow events with faster ascending and descending limbs of the hydrograph), (ii) elevated concentrations of nutrients and contaminants, (iii) altered channel morphology, and (iv) reduced biotic richness, with increased dominance of tolerant species (Walsh et al. 2005). However, most urbanization impacts on streams can be ascribed to a few major large-scale sources, primarily urban stormwater runoff delivered to streams by hydraulically efficient drainage systems. Other stressors, such as combined or sanitary sewer overflows, wastewater treatment plant effluents, and legacy pollutants (i.e., long-lived pollutants from earlier land uses) can obscure the effects of stormwater runoff (Walsh et al. 2005).

The major hydrological modification by urbanization is the transformation of natural land surfaces into impervious surfaces, such as streets, parking lots, roofs and other types of structures that block the infiltration of rainwater and snowmelt into the soil (Shuster et al. 2005; World Water Assessment Programme 2009). The total impervious surface area in the U.S., for example, is approaching the size of Ohio, 1.16×10^6 km^2 (Elvidge et al. 2004). In 2000, the impervious surface cover of the conterminous U.S. was estimated to be 83,749 km^2, and is projected to increase to 114,070 km^2 by 2030 (Theobald et al. 2009). Imperviousness of the soil surface increases, in particular, the flow velocity of water over the land surface and causes drastic decreases in infiltration and increases in surface runoff (Paul and Meyer 2001). Such an urban drainage effect increases the frequency of flash floods, causing loss of human life and damage to infrastructure. For example, increased impervious surface associated with urbanization was primarily responsible for increased frequency and magnitude of high-flow events in urbanized ecosystems within nine metropolitan areas in the U.S. (Brown et al. 2009). However, this pattern was not universal as the metropolitan areas differed in stormwater management and the presence of small reservoirs.

Aside from changing hydrology, impervious surfaces accelerate efflux of pollutants from non-point sources such as buildings, roadways and parking lots into streams (Grimm et al. 2008). Polluting materials are carried into receiving water systems, where they degrade water quality and cause local pollution problems. Thus, increased loadings of nutrients, metals, pesticides, and other contaminants are observed in urban runoff, and in municipal and industrial discharges (Paul and Meyer 2001). For example, urbanization resulted in an increase in specific conductance in streams within nine metropolitan areas in the U.S., and specific conductance was the most consistent indicator of urban stream water quality (Brown et al. 2009). However, effects of urbanization on stream water quality depend also on prior land use. Specifically, effects of urbanization were the least frequently observed in metropolitan areas where agricultural land uses (e.g., crops and pasture) were replaced by urban land uses. Globally, however, a rapid increase in discharge of nitrogen (N) and phosphorus (P) with urban wastewater is predicted from 2000 to 2050 because of the combined effect of increasing population, urbanization, and development of sewage systems (Van Drecht et al. 2009). A strong increase in N and P discharge is predicted specifically for North America despite technological progress resulting in high N and P removals. Between 1970 and 2000, N discharge increased by 39% but P discharge increased only by 1% (Van Drecht et al. 2009).

Urbanization accentuates conversion of natural or agricultural lands to urban soils with altered biological, chemical and physical properties (Lehmann and Stahr 2007). Soil functions particularly important in urban ecosystems are the protection against damages by intense precipitation and flooding, retention and immobilization of contaminants, production of clean water, and buffering of climate extremes, mainly through evaporative cooling (Lehmann 2006). Because of their disturbance by human activities, urban soils have distinct properties which are discussed in the following sections with respect to hydrological fluxes, water quality, and soil organic matter (SOM) content.

4.2 Urban Soils

In contrast to natural soils, human–made materials dominate or strongly influence urban soils as human activities constitute important soil-forming factors in urban ecosystems. Soils whose properties and pedogenesis are dominated by their technical origin are classified as Technosols in the World Reference Base for Soil Resources (IUSS Working Group WRB 2007). Technosols are often referred to as urban or mine soils. They contain large proportions of artefacts (i.e., something in the soil recognizably made or extracted from the earth by humans), or are sealed by technic hard rock (i.e., hard material created by humans, having properties unlike natural rock). Technosols include soils from wastes (e.g., landfills, sludge, cinders, mine spoils and ashes), pavements with their underlying unconsolidated materials, soils with geomembranes and constructed soils in human-made materials. Technosols and their properties have not yet been studied extensively (Lehmann and Stahr 2007). However, a greater understanding of urban soil properties is urgently needed to assess their role in the global carbon (C) cycle and to manage their ecosystem services for the well-being of the urban population (Lorenz and Lal 2009). For example, an increasing number of rural farmers in Africa and Asia move to urban centers, and this increases the demand for food production on urban soils by increasing focus on urban agriculture (FAO 2007). The interactions between urban development patterns and ecosystem dynamics are, however, still poorly understood (Pouyat et al. 2007).

4.2.1 Physical and Hydrological Properties

Soils are characterized by specific physical and hydrological properties including texture, structure, water retention and transmission. Soils have 3 phases and 4 components. Soil physical properties and processes depend on the relative magnitude of phases and components. These phases (i.e., solid, liquid and gaseous) are altered by direct and indirect effects of the conversion of natural/agricultural lands to urban soils. Alterations include physical disturbance, removal or desurfacing, burial or coverage of soil by fill material or impervious surfaces, and soil, water and vegetation management practices (e.g., fertilization, irrigation, mowing, drainage). However, changes in the abiotic and biotic environment by urbanization also indirectly influence soil physical and hydrological properties. For example, the climate is modified, the biodiversity is altered, the soil hydrophobicity is changed, and a range of pollutants are introduced. Transformation of natural to urban ecosystems may, thus, alter soil stratification, soil texture, structure, water retention and transmission characteristics, and probably result in a higher relative proportion of coarse fragments (> 2 mm).

Urban soils are particularly characterized by high variability in bulk density and saturated hydraulic conductivity (Mullins 1991). With increasing anthropogenic perturbations, the air-filled porosity and, thus, the air capacity may increase. The field capacity and available water capacity are, however, often lower in urban than rural soils because of the high contents of coarse fragments. The capacity of urban soils to store plant available water is important since urban soils may be under garden, lawn, park or forest land use. The major direct and indirect effects of urbanization on soil physical and hydrological properties are discussed in more detail in the following section.

4.3 Direct Effects of Human Activities on Physical and Hydrological Properties

4.3.1 Removal and Desurfacing

The physical properties of urban soils show a high degree of horizontal and vertical heterogeneity (Tables 4.1 and 4.2; Baumgartl 1998). In particular, properties such as texture, structure, bulk density, aeration, hydraulic conductivity, water availability, and color may change abruptly at the boundary between two layers in urban soils (Fig. 4.2; Craul 1992). Thus, soil layers exposed to the surface after removing the original cover have a high probability of exhibiting different physical properties (e.g., porosity, erodibility, structural stability, saturated hydraulic conductivity, structure, and rootability) (Mullins 1991). For example, removal of dense/compacted topsoil which occurs in many urban landscapes may expose underlying loosely packed soil horizons. However, the opposite may also occur (Scharenbroch et al. 2005). The bare urban soil exposed after desurfacing may form a crust due to compaction by human and machine traffic, and the low vegetal cover. Raindrops disintegrate soil aggregates and redistribute fine sand, silt, and clay into soil pores leading to a closed packing. Finally, microlayers thus formed are characterized by reduced total volume with attendant decline is size, shape, and continuity of pores (Lal and Shukla 2004). Formation of crust reduces water infiltration and inhibits gaseous exchange with the atmosphere. Removal of surface soil and plant cover

Table 4.1 Soil physical properties of natural and urban soils

Ecosystem	Coarse fraction (w, g kg^{-1})	Bulk density (Mg m^{-3})	Saturated hydraulic conductivity (cm d^{-1})	Field moisture capacity in 0–1 m (cm^{-3})	Plant available water capacity in 0–1 m (cm)
Natural[a]	30–850	0.93–2.00	0.01–30,000	0.13–0.52	6.0–30.0
Urban[b]	0–820	0.71–2.63	1–7,240	0.23–0.37	5.1–14.9

[a]Blume et al. (2010)

[b]Blume and Felix-Henningsen (2009), Jim (1998), Pouyat et al. (2002, 2007), Stahr et al. (2003)

Table 4.2 Hydrological and soil properties along a rural-to-urban gradient in Stuttgart, Germany (Modified from Stahr et al. 2003)

Land use	Depth (cm)	Coarse fraction (w, g kg^{-1})	Bulk density (Mg m^{-3})	Saturated hydraulic conductivity (cm d^{-1})	Field capacity cm^{-3})	Plant available water capacity (cm)	Soil organic carbon (%)
Forest	0–30	68	1.12	1,119		1.50	12.2
	30–100	153	1.63	55		3.57	4.1
	0–100				0.27	10.10	
Agriculture	0–30	32	1.47	165		2.22	15.1
	30–100	47	1.59	221		6.93	4.7
	0–100				0.37	17.30	
Vineyard	0–30	313	1.50	3,194		0.81	11.0
	30–100	423	1.52	3,552		4.69	5.2
	0–100				0.34	9.40	
Park	0–30	106	1.39	1,913		1.05	18.6
	30–100	223	1.30	1,100		6.37	11.6
	0–100				0.33	12.60	
Allotment	0–30	333	1.43	988		0.45	22.7
	30–100	198	1.43	588		6.51	10.5
	0–100				0.37	10.80	
House	0–30	83	1.28	1,098		1.50	26.2
	30–100	111	1.44	1,776		6.93	4.1
	0–100				0.34	14.90	
Village center	0–30	20	1.54	43		1.26	13.4
	30–100	235	1.60	301		4.69	4.1
	0–100				0.33	10.90	
City center	0–30	199	1.43	974		1.11	11.0
	30–100	300	1.63	130		3.22	6.4
	0–100				0.37	8.30	
Road	0–30	795	1.10	n.d.		n.d.	0.6
	30–100	417	1.35	n.d.		n.d.	0.7
	0–100				0.32	8.40	
Railway	0–30	509	1.24	2,846		0.33	21.5
	30–100	616	1.57	1,089		2.24	7.0
	0–100				0.23	5.10	
Military barracks	0–30	492	1.56	140		0.51	5.2
	30–100	533	1.65	21		2.94	3.5
	0–100				0.34	5.90	

may result in the concentration and channelization of surface runoff, or interruption of the natural drainage (Fig. 4.2; Craul 1992). Such alterations accelerate the erosion hazard in urban ecosystems as particle detachment, transport, and subsequent deposition by water are greatly exacerbated. Consequently, the erosion rate in urban ecosystems is much higher than in the surrounding rural/agricultural environments and undisturbed natural ecosystems.

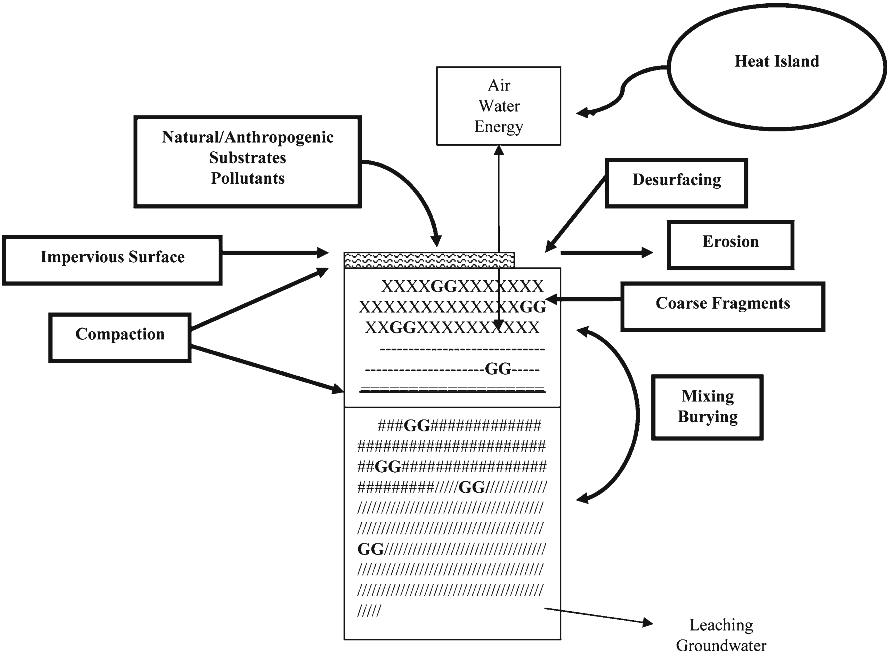

Fig. 4.2 Physical disturbance of a soil profile by urbanization

4.3.2 Burial and Coverage

Soil mixing is the mechanical rearrangement of soil material in situ. Anthropogenic or technogenic materials (e.g., ashes, slag, rubble, bricks, concrete, asphalt, glass, plastic, garbage, or sewage sludge) are often incorporated in urban soils (Craul 1992). Thus, soil structure is perturbed and the depth distribution of soil components is altered (Fig. 4.2). The pore volume and, in particular, the macropores are enlarged in the surface layer whereas deeper soil layers are often compacted (Baumgartl 1998). Thus, soil physical characteristics (i.e., pore size distribution, hydraulic conductivity, water storage capacity, and temperature) are drastically altered. The anthropogenic materials can also dilute the rooting volume, impede root extension mechanically, and mix soil organisms. Mixing of urban soils may also result in the spatial homogenization of soil physical properties.

Anthropogenic substrates, especially the construction wastes, lead to accumulation of a high amount of coarse fragments in urban soils (Table 4.2). In natural soils, gravel is basically physically inert (Lal and Shukla 2004). However, construction wastes and other anthropogenic substrates contain porous materials which have a much greater surface area than non-porous stones (Mullins 1991). These porous materials may, therefore, appreciably enhance water capacity, cation exchange capacity (CEC), and may exert a considerable influence on soil physical properties. Regardless of their nature, coarse fragments have a strong impact on soil physical, mechanical, and hydrological properties. Soil surface properties, such as porosity, rainfall detention and infiltration, moisture distribution, water storage, overland flow, and evaporation are strongly affected. Further, presence of gravelly material alters in the soil profile porosity, water infiltration, percolation and runoff (Mullins 1991).

Covering the soil with impervious surfaces (i.e., concrete, asphalt, or other nonporous material) restricts the exchange of water and gases between the atmosphere and underlying soil horizons (Fig. 4.2; Craul 1992). However, impervious material can adsorb water, release it through evaporation and, thus, be a component of the urban water cycle. Both pore volume and field capacity are reduced, and, in particular, the proportion of medium pores in soils covered by impervious surfaces is smaller. Medium pores are especially important for uptake and transmission of water. Capillary rise, drainage, and evaporation are directly impaired by impervious cover whereas transpiration is only indirectly affected. Increase and acceleration in runoff and reduction in evapotranspiration are, thus, major effects of increase in impervious cover (Nehls et al. 2008). Otherwise, the covered soil may stay moist for long periods of time. Such soil conditions are disadvantageous to root growth and proliferation as aeration and soil moisture regimes are unfavorable. Roads, for example, impact key physical processes such as water runoff and sediment yield (Forman and Alexander 1998). Water rapidly runs off impervious roads and other surfaces as evaporation and transpiration are reduced. An increase in runoff may then accelerate the rates and extent of erosion in connected streams, and reduce percolation and aquifer recharge rates, alter channel morphology, and increase stream discharge rates. Sources of sediments associated with roads are road surfaces, cutbanks, fillslopes, bridge/culvert sites, and ditches (Fig. 4.3). The exposed soil surfaces, as well as the greater sediment-transport capacity of increased hydrologic flows, result in higher erosion rates and sediment yields (Forman and Alexander 1998).

In general, bulk density of surface soil in urban ecosystems may be high because of physical soil modification during construction or texture amendments, use of heavy equipments, and land forming and landscaping of surfaces (Table 4.2; Fig. 4.2; Scharenbroch et al. 2005). Otherwise, subsoil horizons may also be compacted by foundation footers, underground ceiling or sharp changes in physical properties among soil layers (Craul 1992). Soil compaction also decreases the portion of water and air filled soil pores, and the concomitant increase in the portion of the solid phase and increase in soil bulk density. The total pore volume and average pore diameter are reduced and concomitantly bulk density and soil strength are increased. Water holding capacity and air-filled pore space may be reduced. The pore size distribution also changes by compaction causing a loss of total, and especially macropore volume. This reduces water infiltration and gaseous exchange diffusion. The number of water-filled micropores may be increased. Pore pattern and arrangement are also altered which increases tortuosity of the diffusion pathway (Craul 1992).

Fig. 4.3 Exposed soil surface during road construction in Hungary (Snr. László Szalai, http://commons.wikimedia.org/wiki/Main_Page)

4.3.3 Landscaping and Environmental Engineering

Natural hydrologic pathways and processes in urban soils are substituted with engineered drainage and transport systems (Fig. 4.1; Paul and Meyer 2001). The water regime of urban soils can be altered by drainage and irrigation (e.g., lawns, golf courses, trees) to alleviate soil compaction and reduce water-holding capacity together with impeded drainage and high evapotranspiration (Craul 1992). For example, urban grasses in the continental U.S. occupy an area which is three times larger than that of any irrigated crop (Milesi et al. 2005). The amount and rate of water moving into and through the soil profile (i.e., the infiltration) is more sensitive to urbanization than to changes in soil bulk density. Compacted urban soils are characterized by reduced infiltration and permeability. In these soils, the least permeable horizon determines the infiltration capacity of the entire profile. Layers compacted by human activity or concrete application occur in urban soil profiles and alter infiltration capacity. Besides the presence of impervious surfaces, interception from plant canopies in urban settings reduces the amount of water entering the soil. In many urban soils, restricted profile drainage is common, thereby, necessitating internal/surface drainage (Craul 1992).

Urbanization exacerbates fragmentation of wetland habitats, which under natural conditions are inter-connected (Faulkner 2004). Fragmentation of wetland habitats affects the native flora and fauna causing changes in ecosystem functions. Invasive and exotic plants can readily encroach the fragmented wetlands. Changes in the species composition of both the canopy and understory/shrub layers are a common result of urbanization.

4.4 Indirect Effects of the Urban Environment on Physical and Hydrological Properties

4.4.1 Urban Climate

Urban ecosystems impose significant forcing on the weather-climate system because the built environment is a heat source, a poor storage system for water, an impediment to atmospheric exchange and mixing, and a source of aerosols (Hidalgo et al. 2008). Urban aerosols, in particular, have indirect effects on cloud-precipitation microphysics and insolation effects, and on cloud-microphysical and precipitation processes (Seto and Sheperd 2009). Because of the high density of pollutants in the urban heat island plume the regional and global atmospheric climate may be altered (Crutzen 2004). Urban surfaces (i.e., roofs and pavements) have 10–20% lower albedo and, thus, absorb more heat energy than natural/vegetative surfaces (Akbari et al. 2009; Landsberg 1979). Thus, among the familiar manifestations of urban climate modification is the so called “urban heat island” (Jäger and Barry 1990). Trends in urban heat islands in some regions are similar in magnitude or greater than that from radiative forcing of climate by GHGs (Stone 2007; Fujibe 2009). Urban temperatures are generally higher because of changes in the radiation balance, release of heat by human activities, reduction of heat diffusion, and decrease in thermal energy required for evaporation and evapotranspiration. Thus, urban land use can also initiate, modify or enhance formation of the precipitation cloud systems, and create an increase in regional precipitation variability and intensity (urban rainfall effect; Shem and Shepherd 2009). Higher urban air temperatures also cause higher urban soil temperatures (Savva et al. 2010), which in turn may affect soil-water movement and availability, evaporation, and aeration (Lal and Shukla 2004). Furthermore, soil physical properties may also be affected by decrease in radiation, reduction in relative humidity, and decline in wind velocity, and increase in cloudiness and the attendant precipitation in cities.

4.4.2 Biodiversity

Historically, biodiversity studies are rooted in wild or rural landscapes and, thus, environmental impacts of alterations in biodiversity by urbanization are less well known (Dearborn and Kark 2009). For example, urban long-term ecological research

(LTER) sites have been studied merely for a decade in Baltimore and Phoenix, U.S.A.. Nonetheless, urban ecological research sites are non-existent in most cities including those in Europe (Shochat et al. 2006; Pickett et al. 2008; Metzger et al. 2010). Studying natural habitats in urban ecosystems may, however, be beneficial to mitigating adverse effects of climate-change on biodiversity and ecosystem services because of possible increases in temperature, carbon dioxide (CO_2), reactive inorganic N and ozone, factors which are projected to increase in non-urban ecosystems in the coming decades (Carreiro and Tripler 2005).

Urbanization supports high biodiversity because of increase in native and exotic animal and plant communities (Kaye et al. 2006). The fauna of urban green spaces is relatively well studied but those of built-up and derelict areas, and water bodies are less well known (Pickett et al. 2001). Much of the past research on urban fauna, however, is descriptive and qualitative. For example, non-native earthworm species may colonize urban soils or be introduced by human activities, which in turn can alter soil physical and hydrological properties (McDonnell et al. 1997). Also, food webs and their link to ecosystem services may be different in urban ecosystems (Lorenz and Lal 2009). Many cities were established on sites naturally rich in fauna and flora (Kühn et al. 2004). Thus, species diversity in urban ecosystems often decreases as many native species are lost by the dramatic environmental change associated with urbanization and urban land management practices (Hope et al. 2003; Shochat et al. 2010). However, changes in total community diversity may be small as lost species are often replaced by other species such as urban exploiters or urban adapters (Kark et al. 2007). Whether ecosystem services (i.e., C sequestration and buffering of storm runoff) can be achieved with a small set of not necessarily native species in urban ecosystems is, however, not known (Dearborn and Kark 2009).

In this regards, it is important to understand that the role of exotic species in urban ecosystems has not been widely studied (Pickett et al. 2001). Trees in urban ecosystems, for example, often originate from wetlands and floodplains due to lower oxygen tensions in urban soils and predominantly impervious surfaces. Thus, the percentage of native flora decreases from a high in the urban fringes to low levels in the city center. In contrast, exotic plant species are either introduced or encroach because of the habitat fragmentation (Faulkner 2004). The functional role of exotic plants in landscape structure and services in urban ecosystems is, however, not widely studied. In native ecosystems, invasive species can completely alter the hydrology (Mack et al. 2000). In urban settings, exotic trees may affect infiltration of water because of differences in foliation periods, and in crown size and shape compared to those of the native species. Furthermore, exotic plant cover may also differ in the protection against soil erosion compared to soil covered by native species. Litter from exotic species may accumulate on the soil surface because of slower decomposition rates, and, thus, affect both water infiltration and aeration/gaseous exchange. Furthermore, root systems of exotic trees may differ in shape, size classes, and depth distribution affecting soil physical and hydrological properties of the subsoil horizons (Mack et al. 2000).

4.4.3 Pollutants

Pollutants may indirectly affect hydrological and physical properties of urban soils. For example, the deposition of aerosols and particulates which form water repellent compounds may contribute to crusting and surface seal formation in urban soils (Craul 1992). However, inventories of aerosol and particulate emission in cities are often too uncertain to generalize possible alterations in urban hydrology (Fenger 1999). In general, however, pollutants may adversely affect the biotic community in urban soils. The enhanced faunal activity (i.e., earthworms) reduces soil compaction, and increases aggregation, aeration and water holding capacity. Organic matter and minerals are mixed through bioturbation. Yet, coarse fragments are not relocated and may accumulate in a specific layer. Specifically, soil physical properties are altered by the mixing activity of macro- and megafauna (e.g., earthworms, ants, termites). Thus, those pollutants which affect activity and species diversity of the soil fauna may also affect soil hydrological, physical and mechanical properties. In general, urban soils receive more pollutants than agricultural soils (Craul 1992). Also, the rate of fertilizer and pesticide use in urban ecosystems is much higher than those in agricultural and forest ecosystems. In addition, the rate of atmospheric deposition of pollutants is higher than that in natural ecosystems (Craul 1992).

4.5 Urban Hydrosystems

4.5.1 Water Retention

Urbanization can drastically alter the regional hydrology (Fig. 4.1; Tang et al. 2005). Withdrawal of water for residential and municipal uses deplete water resources, and the water balance is modified by urban land use and land cover change. Thus, urban hydrology is drastically altered compared to non-urban rural lands (Pickett et al. 2001). Urbanization increases the impervious surface area cover leading to a decrease in infiltration of water from precipitation and snowmelt (Paul and Meyer 2001). For example, Arnold and Gibbons (1996) observed that 21% of the precipitation in urbanizing watersheds is lost by shallow infiltration with 10–20% imperviousness, and another 21% is lost by deep infiltration. With imperviousness of 75% and 100%, precipitation lost by shallow infiltration is increased by additional 10%, and 5% more is lost by deep infiltration. Thus, urban soils retain less precipitation, as is indicated also by reduced groundwater recharge, and create unique groundwater ecosystems (Grimm et al. 2008). Otherwise, urban forests are important for water conservation where water resources are scant such as those in Beijing, China (Wu et al. 2010). Not only local but also non-local communities in Beijing benefit from watershed protection by urban forests.

4.5.2 Water Transmission

Urbanization has major impacts on the transmission of water in urban watersheds (Fig. 4.1). As impervious surface cover increases, evapotranspiration decreases, and surface runoff increases and urban stormwater runoff is delivered to streams by hydraulically efficient drainage systems (Pickett et al. 2001; Walsh et al. 2005). Therefore, a high proportion of precipitation leaves urban watersheds by surface runoff (Paul and Meyer 2001). Within urban ecosystems, however, evapotranspiration rates are high although highly variable (Arnold and Gibbons 1996). The time of concentration, time difference between the peak of precipitation volume to the peak of runoff volume, is shortened in urban watersheds, resulting in flash floods that peak more rapidly. Furthermore, floods are of shorter duration but peak discharges are higher in urban than in rural watersheds. A reduced groundwater recharge decreases the baseflow discharge in urban streams.

The fragmentation of natural wetlands by urbanization, along with an increase in urban drainage networks, significantly affects soil hydrology and the flood prevention as water is quickly moved away from inhabited areas (Faulkner 2004). Drier and upland forest habitats are more often fragmented by urbanization than wetlands but the hydrologic regime and water quality of downstream wetlands is also affected. Impervious surfaces introduced by urbanization block the infiltration of precipitation resulting in greater surface flow, and channel storm runoff directly into drainage networks (Shuster et al. 2005). Higher peak flows, lower time of concentration to peak flow, more runoff and lower baseflow are among the principal consequences of urbanization. Enhanced transport of water in urban watersheds may also result from stream burial (Elmore and Kaushal 2008), or the direction of streams into culverts, pipes, concrete-lined ditches, or simply covering of streams with pavement. Such alterations are probably the most extreme impacts of urbanization on streams. For example, 66% of all streams within Baltimore City and 70% of streams in watersheds < 260 ha are buried streams. In this densely urbanized city, headwater streams are buried to the same extent as was the dry land (Elmore and Kaushal 2008). However, urban streams and their modifications have not been ecologically studied in the past (Paul and Meyer 2001). Across the urban landscape, local hydrological and erosion effects are dispersed along roads (Forman and Alexander 1998). Roads accelerate water flows and sediment transport, exacerbate flood levels, and degrade aquatic systems.

4.6 Water Quality

Urbanization also affects water quality as is indicated by increased loadings of nutrients, metals, pesticides and other contaminants in urban runoff (Paul and Meyer 2001). When wetlands in urban settings are altered by conversion and habitat fragmentation, their biogeochemical functions which transform and/or retain pollutants are also affected (Faulkner 2004). Nitrate (NO_3^-), for example, is effectively removed in wetlands while P and metals (Pb, Hg, As) are attached to suspended particles, and partially retained through sedimentation. Further, increases in impervious surface areas in urban settings channels pollutants as surface runoff discharges directly into drainage networks and aquatic ecosystems. (NO_3^-) and P concentrations may be elevated in urban runoff, and urban streams may contain high concentrations of these compounds (Paul and Meyer 2001). Urban streams have also high concentrations of ammonium ions. Compared to groundwater sampled in major aquifers in the U.S., (NO_3^-) concentrations in shallow groundwater beneath urban ecosystems are elevated although lower than those beneath agricultural lands (Burow et al. 2010).

Nitrogen losses from urban watersheds are much higher than those from forested watersheds (Groffman et al. 2009). However, total yields of N may be lower from urban than those from agricultural watersheds. Further, N retention in urban ecosystems, specifically in lawns can be high (Raciti et al. 2008). Aside from agriculture, urban activities are major sources of N and P in aquatic ecosystems (Carpenter et al. 1998). Specifically, fertilizers and deposition of reactive N compounds are major source for N in urban water whereas fertilizers, wastewater and erosion are major sources for P (Paul and Meyer 2001; Groffman et al. 2009). Nonpoint sources of pollutants are the major cause of water pollution in the U.S. (U.S. EPA 1996). A significant amount of N and P enters the surface waters from urban nonpoint sources, such as construction sites, runoff of lawn fertilizers and pet wastes, and inputs from unsewered developments (Carpenter et al. 1998). Urban point sources of water pollution, such as sewage and industrial discharges, are also significant sources of pollutants. Construction sites are a critical area of concern for urban nonpoint pollution. Although construction sites may cover a relatively small percentage of an urban ecosystem, erosion rates for construction sites can be extremely high. However, the relationship between N and P concentrations and urban streams depends on other land uses where urbanization is occurring, specifically the agricultural land uses (Brown et al. 2009).

Concentrations of chloride ions are also high in urban streams and groundwater, because chlorides are often the principal deicing salts (Paul and Meyer 2001). Furthermore, metals are exported in runoff and groundwater, primarily in particulate form bound to sediment particles. Urban wetlands have higher levels of As, Cd, Cr, Cu, Ni, Pb, and Zn than rural wetlands. Urban streams are higher in almost all constituents, but especially ammonium, hydrocarbons, and metals because of the runoff from wastewater treatment plants, combined sewer overflow and numerous nonpoint sources (Paul and Meyer 2001). Specifically, industrial discharges, and nonpoint pollution sources (i.e., brake linings, tires, and metal alloys) are major sources of metals in urban waters. A common feature of urban streams is the elevated metal concentrations in the water column. There is a higher concentration of heavy metals (i.e., Cd, Cr, Cu, Hg, Ni, Pb, and Zn) in urban river systems. Higher concentrations of Ag, As, B, Co, Mo, Rb, Sb, Sc, Sn, and Sr are also observed in urban rivers. Pernet-Coudrier et al. (2010) reported a strong reactive potential of dissolved organic matter (DOM) in urban water receiving treated effluent with metals. The high reactivity with organic pollutants is attributed to the dominance of proteinaceous structures in urban DOM displaying a strong hydrophilic nature.

Urban streams are also characterized by a high concentration of pesticides (e.g., insecticides, herbicides, and fungicides). Inputs of insecticides in urban ecosystems are higher than those in agricultural or undeveloped lands. In contrast, inputs of herbicides may be higher in agricultural than in urban lands (Brown et al. 2009). Organochlorine pesticides including dichlordiphenyltrichloroethane (DDT) remain a major concern in urban waters (USGS 1999). The main sources for pesticides in urban watersheds are applications around homes and commercial/industrial buildings but especially in lawns and golf courses. Application rates of pesticides frequently exceed those in agricultural ecosystems by an order of magnitude. Higher pesticide concentrations are also observed in precipitation in urban areas. Frequently detected in urban streams are polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and petroleum-based hydrocarbons (Paul and Meyer 2001). Stormwater runoff is the major route of entry of these organic compounds into urban streams in the absence of industrial point sources. For example, contamination of urban streams and lakes with PAHs is widespread and increasing in the United States. The largest source for PAHs in lake sediments is coal-tar-based pavement sealcoat (Van Metre and Mahler 2010).

Hospital effluent contributes pharmaceutical substances including an array of different chemical compounds into urban streams although data on their toxicity levels are scanty (Paul and Meyer 2001). Aside from pharmaceuticals, a variety of organic compounds from wastewater sources (i.e., additives of personal care products, household chemicals, and industrial chemicals) are also present in low concentrations (Musolff et al. 2010). These micropollutants pose health risks involving endocrine disrupting effects, chemosensitizing effects, possible interactions of contaminant mixtures, and chronic effects by long-term exposure. Thus, urban areas are emitters of micropollutants as the continuous release of treated wastewater from urban watersheds is the most relevant micropollutant source for receiving waters. However, combined sewer overflow can contribute a significant portion of micropollutant loads to the surface waters, and also continuous groundwater discharge may contribute to a considerable part of the micropollutant release from urban sewersheds (Musolff et al. 2010).

4.7 Soil Organic Matter and Water Quality

The SOM is a key determinant of soil physical (soil structure, moisture retention), chemical (ion exchange, buffering) and biological (microbial biomass carbon, soil biotic activity) properties (Waksman 1936). The SOM is a mixture of organic compounds in various stages of decomposition (Kleber and Johnson 2010), and is a major reservoir for plant nutrients (Janzen 2006). In addition to C, H and O, SOM contains N, S and P, exchangeable forms of metals (i.e., Ca^{2+} and Mg^{2+}) and strongly bound complexes of Cu^{2+}, Mn^{3+}, Zn^{2+}, Al^{3+} and Fe^{3+}. Thus, increasing the amount of SOM through soil C sequestration improves soil quality and enhances the net primary production (NPP). It improves the use efficiency of input and enhances other ecosystem services in urban ecosystems (Lal 2007; Lorenz and Lal 2009). Aboveground but, more importantly, belowground inputs of plant litter and the microbial biomass are the major parent materials for SOM formation whereas the faunal biomass is a secondary resource (Kögel-Knabner 2002; Lorenz and Lal 2005). Thus, the amount of SOM is determined by a dynamic equilibrium between inputs from plant, microbial and faunal biomass production and C outputs by mineralization (Kögel-Knabner et al. 2008). Mineralization of SOM is inhibited by stabilization processes such as: (i) selective preservation of recalcitrant SOM compounds, (ii) spatial inaccessibility of SOM to decomposer organisms, and (iii) interactions of SOM with minerals and ions as well as the factors controlling them.

Globally, the SOM pool contains more C than the biotic and the atmospheric pools combined (Lal 2004). However, the conversion of native into urban ecosystems, and the attendant human activities associated with urban land use and management alter the quantity and quality of SOM (Lorenz and Lal 2009). For example, C inputs from NPP are reduced by replacement of vegetation with anthropic urban surfaces such as impervious materials, and other artifacts (Fig. 4.2; Pataki et al. 2006). Further, combustion-derived black C (BC) inputs can enhance the SOM stabilization in urban soils because BC is selectively preserved during mineralization (Lorenz et al. 2006). The stabilization of SOM is also enhanced by its burial at deeper depths during mixing of soil and other construction-related activities (Lorenz and Lal 2005). Therefore, dynamics of SOM are directly affected by other human activities such as import/export of material, soil compaction, sealing, contamination, irrigation, and fertilization. Indirectly, the urban environments favor exotic and invasive animal and plant species, and alter the decomposer community with possible effects on C dynamics in urban soils. The well documented heat island phenomenon and urban atmosphere pollution may also affect the SOM dynamics. Depending on the specific characteristics, soils in urban ecosystems may have higher or lower SOM pools than those in non-urban lands (Lorenz and Lal 2009). However, the magnitude of the SOM pool in urban soils is highly variable in space and time, and ranges between 15 and 285 Mg C ha^{-1} to 1-m depth. Urban soils in the U.S. have the potential to sequester large amounts of SOM-C because of a relatively high belowground to aboveground C ratios and high C densities, especially in residential areas where management inputs and the lack of soil disturbances create conditions for net increases in SOM pools (Pouyat et al. 2006). About 18 petagram (1 Pg = 10^{15} g = 1 Gt) C or 10% of the land C storage in the conterminous U.S. is stored in human settlements, of which 64% is stored in urban soils (Churkina et al. 2010). However, generalizations about the effects of urban environment on SOM quantity and quality are not possible due to the insufficient and erratic data base (Lorenz and Lal 2009).

4.7.1 Nitrogen

Humans have transformed the global N cycle at a record pace due to increase in fossil fuel combustion, growing N demands in agriculture and industry, and pervasive inefficiencies in N use (Galloway et al. 2008). Thus, much of the anthropogenic

N is lost to air, land and water. Major N inputs to urban soils occur during application of N fertilizers and by deposition of reactive N compounds (Lorenz and Lal 2009). For example, about 2–4% of the total N fertilizer in the U.S. is applied to city parks and residential lawns (Ruddy et al. 2006). Similar to agricultural soils, however, maintaining high levels of SOM and rates of C cycling in urban soils enhanced by high plant root and microbial activity are essential to enhance N retention (Drinkwater and Snapp 2007; Groffman et al. 2009). Aside from creating N sinks in SOM, active C cycling also creates N sinks in the urban vegetation. The rapid nitrification of ammonium NH_4^+ released from SOM to NO_3^- in urban soils is modified by anthropogenic alterations of soil pH, aeration, moisture and temperature regimes, and SOM (Lorenz and Lal 2009). The export of N in the form of NO_3^- and NH_4^+ with urban wastewater is, thus, a primary concern in surface water quality (Paul and Meyer 2001; Van Drecht et al. 2009). (NO_3^-) from shallow groundwater is also exported into urban streams (Mayer et al. 2010). Urbanization on forested lands in the U.S. has resulted in increases in total N or $NO_3^- + NO_2^-$ concentrations in stream water with increase in urban intensity (Brown et al. 2009). Also, urban land use change can increase the vulnerability of N retention in a watershed to climate variability (Kaushal et al. 2008). However, agriculture may elevate levels of dissolved N. The latter is not affected substantially by urbanization (Brown et al. 2009).

4.7.2 Phosphorus

Urbanization increases P concentrations in watersheds but the effective increase in P is not as large as that observed for N (Paul and Meyer 2001). In general, both the particle-associated and dissolved P levels are higher in urban than in rural watersheds. Fertilizers applied to lawns and P-based detergents are primary sources for P in urban watersheds (Van Drecht et al. 2009). Further, P stored in urban soils may be mobilized by soil erosion and contribute to eutrophication of receiving waters (Paul and Meyer 2001). The relationship between concentrations of dissolved P and urban intensity is variable as was shown by Brown et al. (2009) for nine metropolitan areas in the U.S.. In general, P concentrations in streams decrease when agricultural lands are converted to urban land because inputs of P in urban lands are lesser than those in agricultural lands. Thus, increases in concentrations dissolved nutrients (N and P) with urbanization are not a general pattern in urban streams, because the concentrations depend partially on other land uses where urbanization is occurring (Walsh et al. 2005; Brown et al. 2009).

4.7.3 Pesticides and Organic Contaminants

Inputs of toxicants to urban streams in the U.S. increased with urbanization (Walsh et al. 2005). Sources of toxicants involving pesticides are lawn and golf course management, and applications of pesticides around homes and other urban buildings

(Paul and Meyer 2001). Insecticides are the most common pesticides in urban streams in the U.S. (Brown et al. 2009). However, concentration of herbicides are not generally high in urban streams. Similar to its effect on dissolved N, agricultural land use elevates concentrations of herbicides to levels that overwhelm an urbanization effect. Yet, urbanization may result in an increase in concentrations of polycyclic aromatic hydrocarbons (PAHs) in streams (Brown et al. 2009). Urban sources of PAHs include combustion processes (e.g., vehicle exhaust) and petroleum-based products (e.g., oil spills, leaking tanks, parking-lot sealcoats).

4.8 Management of Soil Organic Matter to Improve Water Quality

The previous section indicated that maintaining SOM in urban soils enhances the retention of water, nutrients and contaminants. In addition, SOM contributes to detoxification and removal of contaminants (e.g., NO_3^-) by promoting the activity of the biotic community (Groffman et al. 2009). Thus, the quality of water leaving urban hydrosystems can be improved by maintaining or increasing the quantity and quality of SOM levels. For example, Feger (2007) reported that an increase in SOM level of a sandy forest soil within an urban watershed improved groundwater quality and, thus, the quality of the drinking water supply for the city of Mannheim, Germany. Sandy soils, in particular, have limited specific surface area compared to clayey soils, and only store relative small quantities of water and nutrients (Troeh and Thompson 2005). Thus, an increase in SOM in sandy soils is relatively more important for retention and improvement of water quality compared to that in clayey soils. Specific examples of how the water quality in urban ecosystems can be improved by management of SOM are given in the following sections.

4.8.1 Reducing Extreme Hydrologic Conditions

The amount of C stored in urban soils is increased when inputs of soil C exceeds its losses through decomposition, erosion, and subsurface transport into ground water, and C losses associated with urban land use activities (Lorenz and Lal 2009; Post et al. 2009). The primary source of SOM input is plant-derived C and its amount depends on NPP which itself depends on availability of water and nutrients (Kögel-Knabner 2002; Kramer and Boyer 1995). Thus, reducing the extreme hydrologic conditions in urban ecosystems which create unfavorable soil moisture conditions for biotic activity is essential for maintaining and enhancing SOM levels in urban ecosystems. Specifically, surface runoff must be reduced and water infiltration into soils increased. Increase in soil water storage can be achieved by reducing the impervious urban surface cover. Specifically, sealings by cobblestones and concrete slabs with open seams can be used as cover instead of impervious sealing material (Nehls et al. 2006). Water infiltration can also be increased by increasing the

Fig. 4.4 Chicago city hall green roof (TonyTheTiger, http://commons.wikimedia.org/wiki/Main_Page)

proportion of multistory buildings which decrease the need for conversion of open urban areas covered by vegetation and soil into single-story buildings. Urban lawns and forests are particularly important for increasing water infiltration and creating water-sensitive urban design (Vernon and Tiwari 2009). Restoring degraded urban soils, such as those in brownfields, is advantageous to improving soil water infiltration and the water holding characteristics for establishing an effective vegetation cover and enhancing accumulation of SOM. Increasing urban vegetation cover and albedo can mitigate or reduce the urban heat island effect (Grimm et al. 2008), which influences urban water resources by changing the surface-energy balance, altering both heat and moisture fluxes near the surface, and changing soil moisture content.

Reductions in extreme hydrologic conditions in urban ecosystems can also be achieved by establishing green roofs on buildings (Fig. 4.4; Oberndorfer et al. 2007). Green roofs are an effective tool for managing small storms in highly developed urban areas (Berndtsson 2010). Substrates/soils of green roofs can be managed for enhancing the organic C/SOM pool, water retention and improving urban drainage water quality. Stormwater runoff reduction by green roofs, for example, can be as much as 27% and 87% of annual precipitation (Berndtsson 2010). A significant proportion of precipitation is retained in the green roof, which either evaporates or is used by plants and transpired as green water. Subsequently, water returned from green roofs back to the urban atmosphere by evapotranspiration can precipitate on other vegetated urban surfaces and, thus, contribute to an increase in SOM levels and improvement in water quality (Oberndorfer et al. 2007). Also, runoff peaks from green roofs are lowered and delayed, and plant growth on the roof contributes to an increase in SOM content of the urban ecosystem (Berndtsson 2010). However, Köhler and Poll (2010) observed in Berlin, Germany, that the organic C content of the green roof substrate approached a steady state level in about 20 years.

There are few if any studies about runoff water quality from green roofs. Some green roofs, for example, do not release P in runoff. However, recently established green roofs and those established with soil enriched with nutrients, and fertilized green roofs can be P sources (Berndtsson 2010). Fertilization, soil type and roof age may also affect concentrations of N in green roof runoff. While (NO_3^-) is generally retained in green roof soil and vegetation, N in the form of (NO_4^+) in roof runoff is reduced compared to precipitation. However, organic N may be released from green roofs as total N concentrations may be similar in precipitation and green roof runoff (Berndtsson 2010). Further, improvements in water quality comprise generally lower concentrations of heavy metals in roof runoff than those in urban runoff from hard surfaces. Green roofs also mitigate acidity of rain water as pH values in green roof runoff water are elevated towards more alkaline values (Berndtsson 2010).

Establishment of water-sensitive urban designs can contribute to water conservation and improvement in the water balance and quality in urban ecosystems (Vernon and Tiwari 2009). Types of stormwater management practices include vegetative controls (overland flow and grassed channels), detention (wet basins and wetlands) and retention (infiltration basins). Vegetative controls, for example, improve water quality and maintain groundwater recharge as the grass vegetation prevents erosion, enhances settling of suspended soil particles and filters pollutants from runoff. Constructed wetlands incorporate large quantities of stormwater runoff and associated materials, comprising either of dissolved pollutants or suspended products. Treatment or removal of pollutants in wetlands occurs primarily through sediment retention and biological uptake. An infiltration basin is used to temporarily store stormwater runoff, thereby allowing it to infiltrate into the ground, and increase groundwater recharge (Vernon and Tiwari 2009). Thus, urban stormwater management can reduce the extreme hydrologic conditions in cities and contribute to maintaining or enhancing SOM levels, and improving water quality (Grimm et al. 2008).

4.8.2 Closing Nutrient and Water Cycles

The SOM in urban soils can be managed and levels increased by closing the nutrient and water cycles within urban ecosystems. Urban centers accumulate N and P which increases uptake by urban vegetation (Kaye et al. 2006). In this context, Grimm et al. (2008) proposed to use (NO_3^-) rich groundwater to fertilize lawns in Phoenix, AZ. The reduction of external nutrient inputs, reuse of city wastes and wastewaters with their nutrients can also be achieved by urban agriculture (De Bon et al. 2010). It is important to realize, however, that recycling of nutrients accumulated in cities may not be adequate to achieve high NPP and agronomic production. Urban agricultural production emerged simultaneously with the unprecedented growth of cities and includes aquaculture, livestock and plants. The world's largest urban farm, for example, is planned in Detroit, MI (http://www.hantzfarmsdetroit.com/). Direct improvements of SOM levels by using organic matter and organic wastes for urban agriculture in developing countries is, however, not widespread. Also, human health concerns need to be addressed when applying urban wastewater. Other constraints to the development of urban agriculture are polluted soils, water resources and air, and the heavy use of pesticides (De Bon et al. 2010). Thus, organic agriculture and phytoremediation have been proposed to reduce the risk of pollution of water, and solid wastes used for compost and soil, with pesticides and heavy metals. Through a judicious management of urban vegetation, nutrient and water cycles can be closed by reducing use of nutrient, water and pesticide, and planting low-input vegetation adapted to the local urban environmental conditions.

Urban streams must also be managed to retain organic matter and nutrients (Groffman et al. 2005). Specifically, geomorphic structures rich in organic matter such as sediments below the urban stream surface, gravel bars next to the stream, or organic debris dams in the middle of the stream are important sinks for (NO_3^-) due to high denitrification enzyme activity (Mayer et al. 2010). At the groundwater-surface water interface, the availability of dissolved organic carbon (DOC) limits denitrification and removal of (NO_3^-). Thus, increasing availability of DOC to denitrifiers, reducing stream-flow velocity and flashiness, increasing time of concentration, and increasing the mean residence time of groundwater will likely improve the N removal capacity of urban stream channels (Mayer et al. 2010).

4.8.3 Enhancing Soil Organic Matter

Urban soils must be managed to maintain and enhance the SOM level for long-term provision of ecosystem services such as nutrient, water and contaminant retention. Increase in SOM pool can be achieved by maintaining and increasing soil C inputs and soil C residence time (Post et al. 2009). In addition to the amount, the nature of C inputs to urban soils can be managed to enhance the SOM pool. For example, applying biochar to urban soils may be an effective strategy to enhance the SOM pool as its characteristic soil storage time is in the millennial range and, thus, substantially greater than that of non-charred biomass (Warnock et al. 2007). In addition to improving nutrient reserves, enhancing water and contaminant retention, soil application of biochar may also promote NPP and, thus, increase inputs of soil C (Sohi et al. 2010). The stability of biochar is related to its poly-condensated aromatic structures. Biochar is produced when organic materials (i.e., wood chips and wood pellets, tree bark, crop residues, energy crop, organic wastes, chicken litter, dairy manure or sewage sludge) are thermally decomposed during slow pyrolysis at low-moderate temperatures and long heating time under limited supply of oxygen (Sohi et al. 2010). In cities, organic wastes and other organic materials available (i.e., lawn cuttings and biomass from urban forests) are potential sources of biochar production (de Richter et al. 2009). In addition to biochar and BC, however, interactions with the soil mineral phase and physical protection are more important to the long-term stabilization of C in SOM than the chemical composition of C inputs in soil (Marschner et al. 2008). Minimizing physical disturbance of urban soils is also an effective way to maintain SOM and enhance its physicochemical protection. This action is similar to the reduced disturbance of crop soils by no-till on conservation agriculture (Kassam et al. 2009).

As discussed in the previous section, enhancing the downward vertical spread of organic C through the urban soil profile has the benefit of long-term SOM protection (Lorenz and Lal 2005). Inputs of organic-C into subsoils occur as DOC is illuviated through preferential flow pathways (by-pass or macropore flow), as aboveground biomass or root litter and exudates are transported along root channels and/or through bioturbation (Rumpel and Kögel-Knabner 2011). However, the relative importance of these processes to enhancement of SOM in sub-soil is not known. Thus, it is unclear whether the SOM pool in deeper layers of urban soils can be maintained or increased through practices such as surface application of organic residues releasing high amounts of DOC, by increasing urban vegetation cover with deep extended root systems and a high root turnover, and by creating favorable conditions for faunal bioturbation (Lorenz and Lal 2005).

The effectiveness and relative importance of various SOM stabilization mechanisms depends on soil type and other conditions (von Lützow et al. 2006). However, management-induced increases in profile SOM levels may ultimately slow down and the C-sink capacity of urban soils may get saturated (Stewart et al. 2008). While the unique biological, chemical and physical properties of an urban soil profile may define an overall saturation limit, disturbances by urban land use and management may define a lower effective saturation or soil-C sink capacity (Stewart et al. 2007; West and Six 2007).

4.9 Conclusions

Urbanization drastically disturbs natural hydrosystems specifically by increasing the impervious cover. The latter decreases water infiltration rate and increases surface runoff. In addition the amount of water, the quality of water leaving urban ecosystems is also affected by increased loadings of nutrients, metals, pesticides, and other contaminants in urban runoff, municipal and industrial discharges. However, increasing the urban area covered by vegetation and soil can increase retention and quality of water. Specifically, increasing the levels of SOM by increasing the pervious sealing cover types, establishing green roofs on buildings and restoring degraded urban soils are management options to reduce urbanization effects on local and regional hydrology and water quality.

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Part II
Urban Forests

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

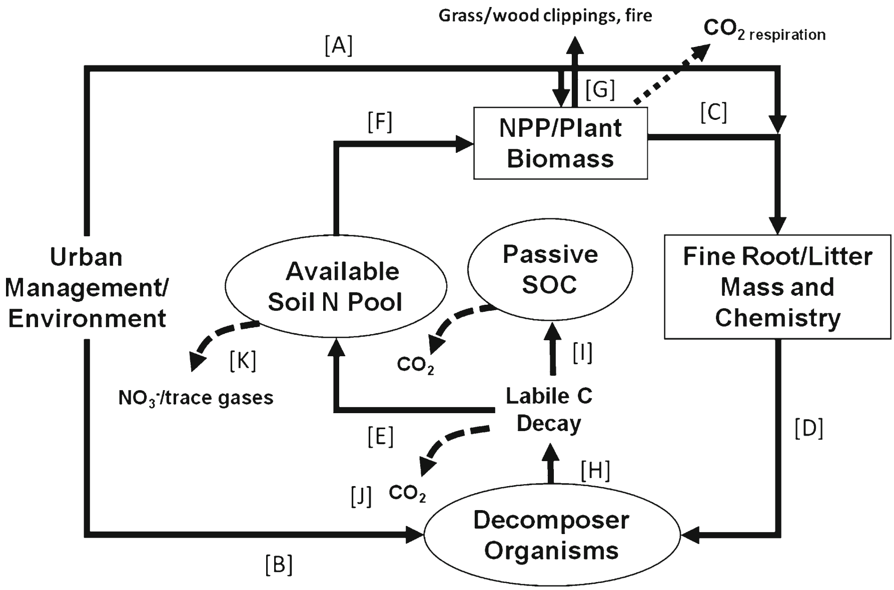

Fig. 5.1 Conceptual diagram of urban environmental change and urban landscape management effects on carbon cycling. Processes are represented as capital letters and explained in the text. *Dashed lines* represent gaseous losses from the ecological system, *solid lines* represent inter connective relationships 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.

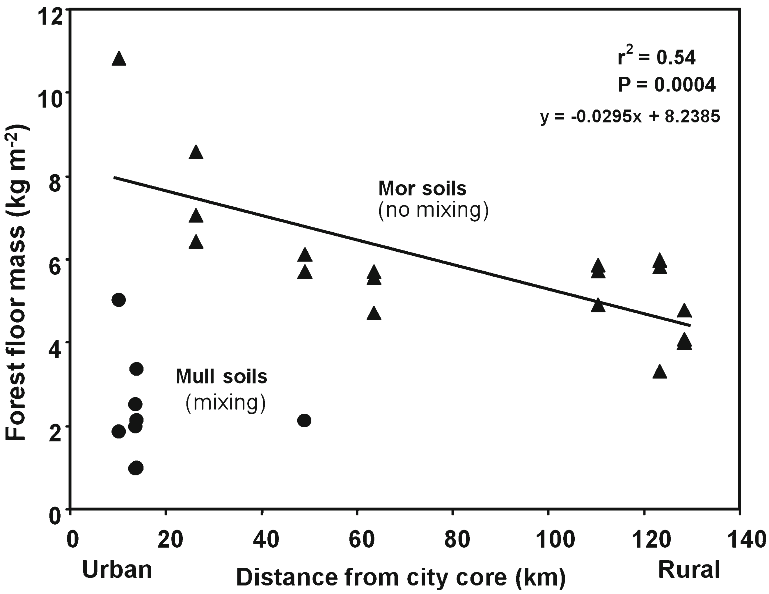

Fig. 5.2 Forest-floor mass of a New York City urban-rural gradient comparing mor soils (*triangles*: no mixing of the O and A horizon) to mull soils (*circles*: mixing of O and A horizon) (Modified from Pouyat et al. 2002)

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 in the native soil replaced (Pouyat et al. 2006, 2009a). The primary cause for increasing C storage in what were once disturbed soils, is the addition of water and nutrients, which under native soil and climate conditions could otherwise be severely limiting NPP. Therefore, in climates supporting low NPP, the input of plant biomass for decomposer organisms also is resulting in low soil C pools (C in Fig. 5.1). With the introduction of supplements such as irrigated water and fertilizer, NPP increases more than decay rates with the potential to increase the soil C pool. This increase is even greater if a portion of the NPP is not removed from the system by practices such as mowing, raking of leaves, and pruning of dead and live tree branches (Nowak and Crane 2002). Therefore, any management practice that returns a portion of NPP to the soil system should result in greater accumulations of C in urban soils. The importance of management practices in accumulating C in urban soils suggests that from parcel to parcel, the variability in soil C pools is strongly dependent on the 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 cycles is the replacement of native cover types with lawn cover (Kaye et al. 2005; Milesi et al. 2005; Golubiewski 2006; Pouyat et al. 2009b). The estimated amount of lawn cover for the conterminous U.S. is 163,800 ± 35,850 km^2, or 73% of all irrigated cultivated lands (excluding lawn cover) (Lubowski et al. 2006). With respect to turf-grass cover, it is estimated that roughly half of all residences apply fertilizers (Law et al. 2004; Osmond and Hardy 2004), some apply fertilizer at rates similar to or exceeding 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 high, urban soils can accumulate a surprising amount of C and N compared to agricultural or native soils (Fig. 5.1). Horticultural management, e.g., fertilization and irrigation, tend to maximize plant productivity and accumulation of soil organic matter for a given climate or soil type and thus increase the capacity of these soils to store C and N. This is particularly true of turf-grass systems in which soils are not cultivated regularly and turf-grass species typically grow through an extended growing season relative to most native grassland, forest, and crop ecosystems (Pouyat et al. 2002; Groffman et al. 2009). The total C budget for maintaining lawns should include emissions resulting from mowing, irrigating, and fertilizing lawns (Pataki et al. 2006).

The literature suggests that turf-grass systems can accumulate C to levels that are comparable to or exceed other grassland and forested systems. Qian et al. (2003) used the CENTURY model to show that N fertilization coupled with replacement of grass clippings increased soil C by as much as 59% compared with sites that were not fertilized and where clippings were removed. Likewise, in measuring soil C stocks in residential areas in the short-grass prairie of the Colorado Front Range, Golubiewski (2006) found that soil C in turf-grass areas exceeded that in semi-arid 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

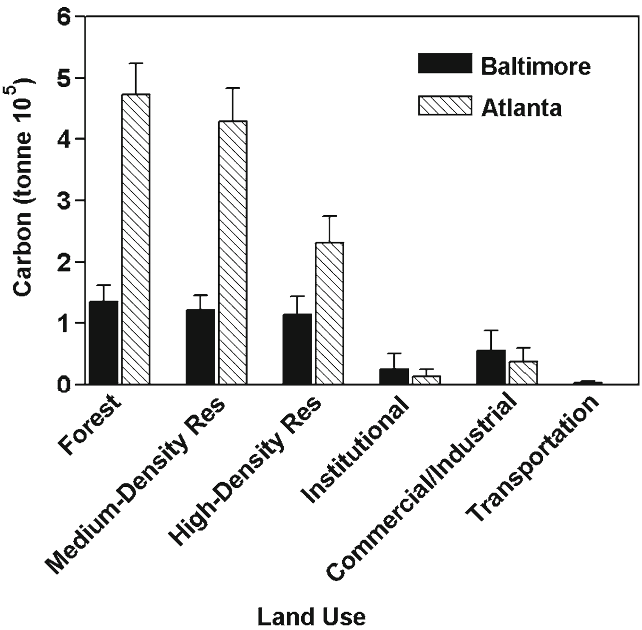

Fig. 5.3 Mean (S.E.) of total carbon storage (10^5 tonnes) in trees for *Baltimore* and *Atlanta* by land use (Data source: Nowak and Greenfield 2009)

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

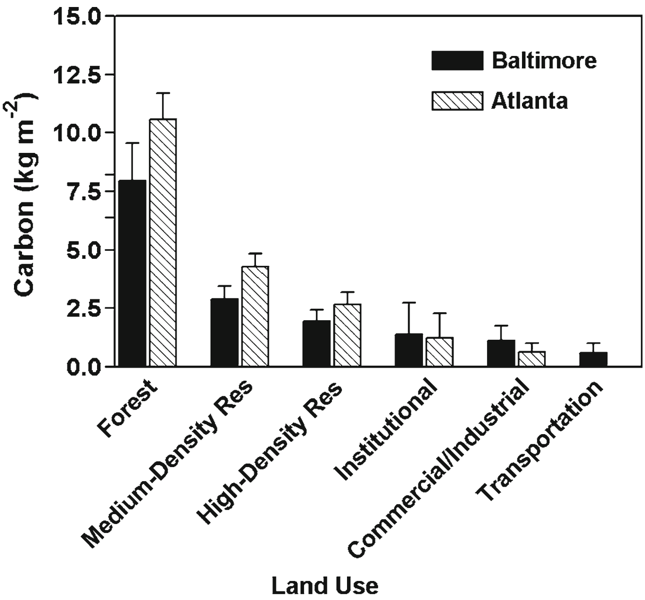

Fig. 5.4 Mean (S.E.) of carbon density (kg m^{-2}) of trees for *Baltimore* and *Atlanta* by land use (Data source: Nowak and Greenfield 2009)

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).

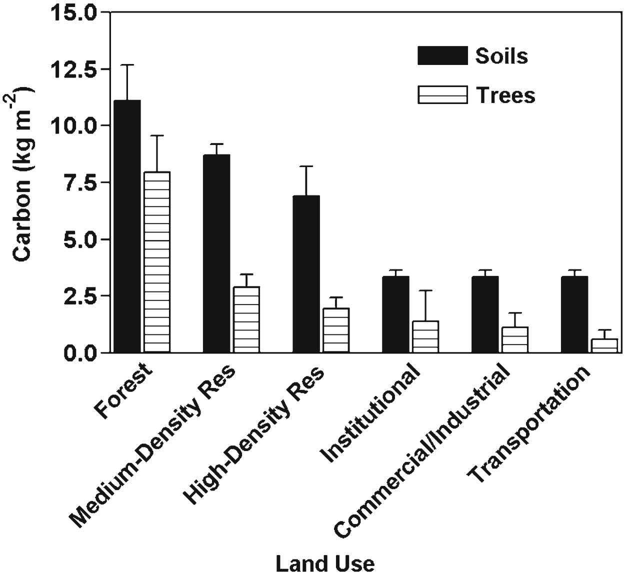

Fig. 5.5 Mean (S.E.) of carbon density (kg m^{-2}) (both pervious and impervious) of *soils* and *trees* in Baltimore by land use (Data source: Nowak and Greenfield 2009; Pouyat and Yesilonis, unpublished)

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.

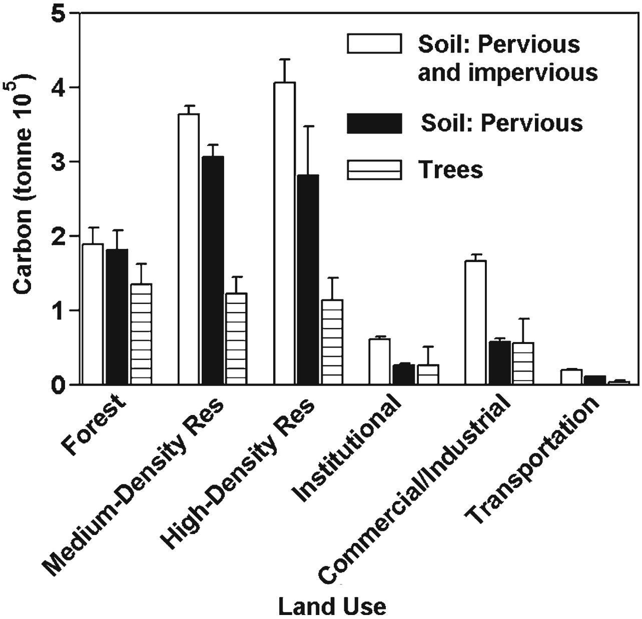

Fig. 5.6 Mean (S.E.) of total carbon storage of trees and soils (10^5 tonnes) for Baltimore by land use. Soils are divided by those under pervious surfaces only, i.e., no impermeable layer above the surface horizon, and those under pervious and impervious surfaces. The latter include soils under pervious 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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Chapter 6
Urban Trees for Carbon Sequestration

B.C. Scharenbroch

Abstract Urban trees represent potential carbon (C) sequestration pools. To examine the C sequestration of an urban tree, factors such as C storage in biomass, avoided C emissions from energy conservation, and C emissions associated with tree maintenance and decomposition must be considered. Maximum tree sizes, lifespans, growth rates, and tolerances to urban stress were used to create four metrics for comparing the relative C sequestration potential of 145 popular urban tree species. These analyses show that differences in the relative urban tree C index (RUTCI) may be related to phylogenetic characteristics. In particular, longer-lived species with higher tolerances to urban stresses and higher wood densities appear to have the highest potential for sequestering C. Current urban tree populations are comprised of species with moderate ratings according to the RUTCI, suggesting that C sequestration potential of urban forests can be improved.

Keywords C storage • C emission • Tree biomass • Tree lifespan • Tree growth rate • Urban stress

6.1 Introduction

Estimates of carbon (C) storage in 17 urban forests in U.S.A. and Canada range from 18,144 to 1,224,700 Mg of C or 4.9 to 37.7 Mg ha^{-1} (Nowak et al. 2010). Carbon sequestration in these urban forests ranged from 494 to 42,093 Mg C and 0.202 to 1.233 Mg C ha^{-1} $year^{-1}$ (Nowak et al. 2010). Mean C storage and sequestration in these urban forests is 418,746 Mg C, 21.0 Mg C ha^{-1}, 14,284 Mg C $year^{-1}$, and 0.67 Mg C ha^{-1} $year^{-1}$ (Nowak et al. 2010). Total urban area in the United States is estimated at 24.1 million hectares or Mha (U.S. Census Bureau 2002), so total

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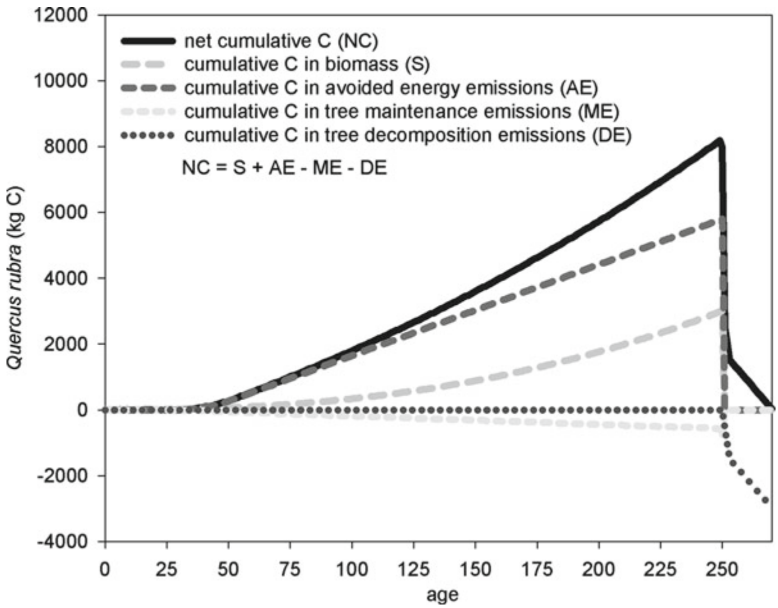

Fig. 6.1 Model of net cumulative C (*NC*), C in biomass (*S*), C in avoided energy emissions (*AE*), C in tree maintenance emissions (*ME*), and C in tree decomposition (*DE*) for a 250-year old *Quercus rubra* with intensive management and mulching decomposition. Biomass C predicted from allometric equation from Jenkins et al. (2003) with 0.8 urban factor (Nowak 1994), 1.2 root biomass factor (Cairns et al. 1997), 0.5 C conversion (Koch 1989). Avoided energy emissions modeled based on Nowak et al. (2002), 27.7 kg C year^{-1} for a 7.6 m tree. Intensive management is planting with a backhoe, pruning every 7 years with light-duty and heavy-duty chainsaws, an aerial lift, and removal with similar equipment, including a stump grinder, upon its death in year 250. Mulching decomposition releases 50% of stored C in first 3 years and the remaining 50% over the next 20 years (Scheu and Schauermann 1994) Net cumulative C is zero in year 270, NC = S + AE − ME − DE

urban forest C storage is 506 million Mg C (Tg C) and annual C sequestration is 16.15 Tg C year^{-1}. Others estimate the C storage U.S.A. forests to be 350–750 Tg of C (Nowak and Crane 2001). The C storage in non-urban U.S.A. forests ranges from 36 to 203 Mg C ha^{-1}, with a mean value of 77 tons C ha^{-1} (Smith et al. 2006). Birdsey and Heath (1995) estimate 4.4% of the total U.S.A. forest C storage is in urban trees. Estimates of U.S.A. greenhouse gas or GHG (CO_2, CH_2, and N_2O) emission range from approximately 1,500–1,700 Tg C year^{-1} (U.S. EPA 2009). The annual C sequestration in U.S.A. urban forests is currently only about 1% of total U.S.A. annual GHG emission. However, urban forest C storage and sequestration may become more important as the urban lands increase. Currently, nearly 80% of the U.S.A. population resides in urban areas, and the land area dedicated to urban use continues to expand (Acevedo et al. 2006).

The net C storage potential of an urban tree should include C stored in biomass, the avoided emissions associated with energy reduction, emissions from tree maintenance, and the emissions from tree disposal (Nowak et al. 2002). A model is presented with these four components of net C storage for a 250-year old red oak (*Quercus rubra*) (Fig. 6.1). Storage of C in biomass is typically computed using allometric equations to predict tree biomass from field measurements such as diameter or height (e.g., Ter-Mikaelian and Korzukhin 1997; Jenkins et al. 2003). Conversion factors of 0.48–0.56 may be used to convert fresh to dry biomass (Monteith 1979). Conversion factors of 1.22–1.26 can be applied to include belowground biomass (e.g., Cairns et al. 1997). To adjust for urban factors (e.g., pruning, compacted soils, etc.), 0.75–0.90 adjustments are often applied for urban tree biomass (Nowak 1994; McHale et al. 2009). Tree biomass is converted to C by applying a 0.5 factor (Koch 1989).

In the northern hemisphere, urban deciduous trees on south and westerly aspects of buildings potentially reduce energy emissions by shading summer sun (McPherson et al. 1993; McPherson 1994; Nowak 1994). Coniferous trees may reduce emissions associated with winter heating by blocking winter winds. The average annual savings of a 7.6 m urban tree on a one-story residential home in cooling is +784.8 MJ at 0.044 kg C MJ^{-1} and in heating is −527.5 MJ at 0.014 kg C MJ^{-1}. The net C emission avoidance for that tree is 27.1 kg C $year^{-1}$ (Nowak et al. 2002). However, urban trees may also increase C emissions from buildings. The net C avoided emissions from Chicago's urban forest in 2008 was −1,200 Mg C $year^{-1}$, −3,600 Mg C $year^{-1}$ from increased C emissions with heating and +2,400 from C emission reductions in cooling savings (Nowak et al. 2010).

Emissions of C from tree maintenance occur with tree planting, pruning, and removal activities. Transportation C emissions are calculated based on fuel efficiency and distances traveled. A 0.7 kg C L^{-1} of fuel use factor can be applied to compute the C emissions from transportation (U.S. EPA 1991). Other types of equipment associated with tree maintenance may include: aerial lifts, backhoes, chainsaws, chippers, and stump grinders. Emissions of C per specific piece of equipment can be computed from hours of use, horsepower, load factors, and the average C emission per unit use (Graham et al. 1992; Murrell et al. 1993; Davis 1994). The total C emission for tree maintenance is dependent upon the intensity of management. For instance, maintenance C emission for a red oak, planted with a backhoe, pruned every 7 years with light-duty and heavy-duty chainsaws, an aerial lift, and removed with similar equipment, including a stump grinder, upon its death in year 250 would be 342 kg C. In comparison, the C emission for that same red oak without pruning and heavy equipment at planting would be 50 kg C. The red oak would need to live 101 and 50 years, respectively, to balance those maintenance C emissions with C storage in biomass (Fig. 6.1).

Upon tree death, biomass C is emitted back to the atmosphere or transferred to long-term C storage pools such as soil or wood products. When tree disposal involves chipping and mulching of woody tissues, it is estimated that 50% of biomass C is lost to the atmosphere in the first 3 years and the remaining 50% is returned to the atmosphere over the next 20 years (Scheu and Schauermann 1994; Nowak et al. 2002). Under this scenario, the modeled 250-year old red oak would be net C sink for 250 years and then a C source over the next 20 years (Fig. 6.1). However, this disposal C estimate does not include the C that may be transferred to long-term soil C storage pools. Soil organic matter (SOM) models accept C inputs from plant restitution and depict C storage pools of varying turnover times ranging from years to centuries (Parton et al. 1994). With a landfill disposal scenario, Micales and Skog (1997), estimate that 4% of woody biomass is released to the atmosphere in the first 5 years when placed in landfills. Urban wood utilization (e.g., Illinois Emerald Ash Borer Wood Utilization Project) is receiving more attention as of late and the manufacture of wood products from urban trees will likely extend the time in which C remains stored in biomass.

Nowak et al. (2002) suggested that urban trees with longer lifespans and larger maximum sizes have greater potential to sequester more C relative to shorter-lived, smaller trees. However, urban constraints, such as soil compaction, interrupted nutrient cycling, light and water availability, salts, etc. may limit lifespan and maximum tree sizes (Craul 1999; Harris et al. 1999; Scharenbroch and Lloyd 2006). Consequently, tolerance to urban stress agents must be considered when evaluating an urban tree's ability to sequester C. The first objective of this research was to create four metrics allowing for the comparison of relative C sequestration potential of 145 common urban trees. Next, species functional groupings were examined to identify the characteristics that differentiate an urban tree's potential to sequester C. Lastly, species compositions of current urban forests were assessed in the context of their potential for C sequestration.

6.2 Methods

A total of 145 popular urban tree species were evaluated using four metrics relating their relative potential to sequester C. The last positive point (LPP), relative urban stress tolerance (RUST), and maximum C storage (C_{st}) were computed for each tree species. The average weighted means of these three metrics were then summed to compute the relative urban tree C index (RUTCI). Species groups for analyses were assigned following Jenkins et al. (2003), which are based on phylogenetic relationships and other factors related to data availability.

Assuming fossil fuel use with tree maintenance, the last positive point (LPP) is the point at which C emission exceeds sequestration for a given tree species planted continuously at a given site (Nowak et al. 2002). The LPP is dependent upon C storage and emission in relation to tree size, lifespan, growth rate, and maintenance activities. Tree sizes, lifespans, and growth rates were identified from the literature (Collingwood and Brush 1964; Hightshoe 1978; Clark 1985; Dirr 1990) and the LPP was assigned based on data provided in Nowak et al. (2002).

Nine potential urban stress agents were assigned a numeric value from −1 to 1, and then summed to compute the RUST. Hardiness and pH ranges were calculated by dividing the range for the species of interest by the maximum range given for the listed 145 species. Sun, insect/disease, physiological/environmental, moisture, salt, texture, and compaction tolerances were calculated by assigning values of −1 for intolerance, 0 for intermediate or not listed, and 1 for an identified tolerance (Collingwood and Brush 1964; Hightshoe 1978; Clark 1985; Dirr 1990; Flint 1997). A numeric value was assigned for each component of the individual tolerance measure. For instance, a tree that was listed as tolerant to insect, but intolerant to diseases, would receive a 0 score, −1 for insect and +1 for diseases. Another example is for a species listed as tolerant to sand, intolerant to clay, and impartial to silt, receiving a score of 0 for texture tolerance.

The C_{st} for each species was computed using allometric equations listed in Jenkins et al. (2003), biomass = exp ($B_0 + B_1$ * ln dbh), with B_0 and B_1 for each species. When species data was not listed, genus was used to predict biomass. Maximum diameters for each species were assigned as the lower of the values listed in Jenkins et al. (2003) and American Forests (2010). Tree lifespans were estimated from Hightshoe (1978), Clark (1985), and Dirr (1990). An urban tree factor of 0.8 was applied to reduce urban tree biomass relative to non-urban trees (Nowak 1994). A factor of 1.2 was applied to include the belowground component (Cairns et al. 1997). Biomass was converted to C content using a 0.5 factor (Koch 1989). Sequestration of CO_2 (C_{seq}) (kg CO_2 year^{-1}) was computed for each species by multiplying C_{st} by 3.664 (44.0098/12.011, molecular weights of CO_2 and C, respectively) and dividing by the tree lifespan. Tree specific gravity was compiled from Haygreen and Bowyer (1996).

Urban tree inventories of four cities in the Chicago, IL area were assessed to examine species distributions in relation to C sequestration potential. Data from Chicago, IL was attained from the Nowak et al. (2010). Data sets from Elburn, IL, Lincolnwood, IL, and Oak Park, IL were collected from randomly located field plots measured by M. Duntemann of Natural Path Urban Forestry Consultants (Duntemann, 2010, Data from urban forest inventories of Elburn, Lincolnwood, and Oak Park, IL, personal communication).

Statistical analyses were conducted using SAS JMP 7.0 software (SAS Inc., Cary, NC). Data distributions were checked for normality using the Shapiro-Wilk W test. Species groups were assigned as treatments and the metrics (RUST, LPP, C_{st}, and RUTCI) were analyzed using analysis of variance (ANOVA), with mean separations carried out with Tukey-Kramer HSD tests, or Kruskal-Wallis tests for nonparametric data. Multivariate correlations and principal component analyses (PCA) were used to explore relationships among the measured variables. The Spearman's p test was used to examine nonparametric correlations. Significant differences were determined at the 95% confidence level.

6.3 Results and Discussion

Species with the highest LPP scores included: pond cypress (*Taxodium ascendens*), bald cypress (*Taxodium distichum*), red horsechestnut (*Aesculus × carnea*), black walnut (*Juglans nigra*), sweetgum (*Liquidambar styraciflua*), London planetree (*Platanus × acerifolia*), American sycamore (*Platanus occidentalis*), scarlet oak (*Quercus coccinea*), red oak (*Quercus rubra*), live oak (*Quercus virginiana*), ponderosa pine (*Pinus ponderosa*), red pine (*Pinus resinosa*), white pine (*Pinus strobus*), and Douglas fir (*Pseudotsuga menziesii*) (Appendix). American holly (*Ilex opaca*), red elm (*Ulmus rubra*), sweetbay magnolia (*Magnolia virginiana*), American elm

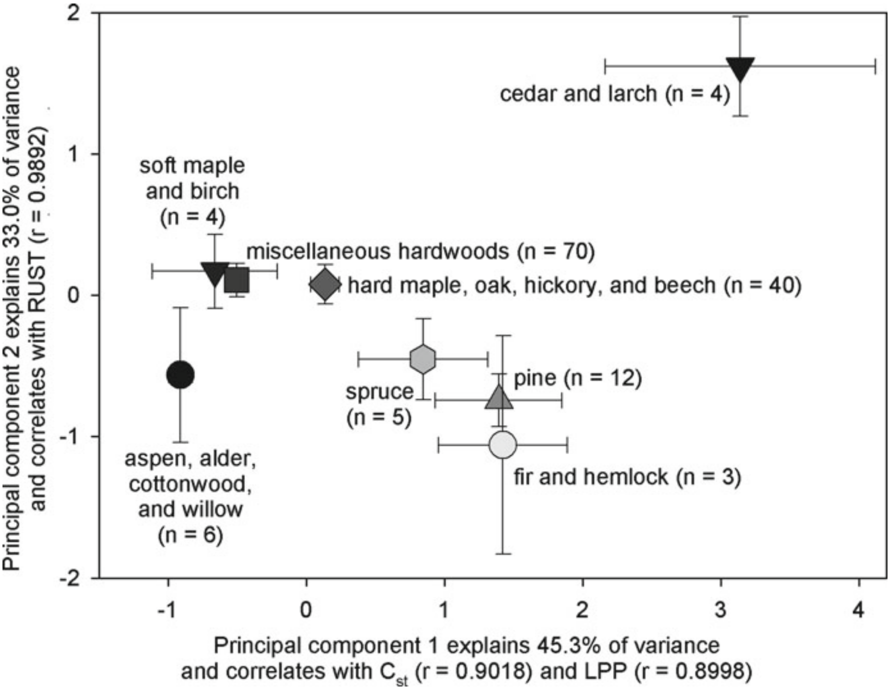

Fig. 6.2 Principal component analysis for maximum C storage (C_{st}), last positive point (LPP), and relative urban stress tolerance (RUST)

(*Ulmus Americana*), Kentucky coffeetree (*Gymnocladus dioicus*), hedge maple (*Acer campestre*), shagbark hickory (*Carya ovate*), shingle oak (*Quercus imbricaria*), bald cypress, sweetgum, and swamp white oak (*Quercus bicolor*) had the highest RUST scores (Appendix). The following species all had>5 Mg C storage (C_{st}) potential: bald cypress, pond cypress, mugo pine (*Pinus mugo*), ponderosa pine, white, pine, northern white cedar (*Thuja occidentalis*), eastern hemlock (*Tsuga Canadensis*), Colorado blue spruce (*Picea pungens*), Norway spruce (*Picea abies*), loblolly pine (*Pinus taeda*), and Scots pine (*Pinus sylvestris*) (Appendix). Both C_{st} and LPP were negatively correlated with RUST scores ($r=-0.0401$ and -0.1569), but neither of these correlations were significant ($p \geq 0.0603$). A significant positive relationship was observed for C_{st} and LPP ($r=0.3970$ and $p<0.0001$). Species that have relatively higher tolerances for urban stress do not appear to the species predisposed with physiological traits for optimal C sequestration (i.e., longer lifespan and larger sizes).

The first and second principal components explained 45.3% and 33.0% of C_{st}, LPP, and RUST variance (Fig. 6.2). The first principal component was correlated with C_{st} and LPP (r values of 0.9018 and 0.8998), respectively (Fig. 6.3). The second principal component was correlated with the RUST scores, $r=0.9892$ (Fig. 6.2). The cedar and larch group was positively correlated with both components, thus higher LPP, C_{st}, and RUST scores. The spruce, pine, and fir and hemlock groups had posi-

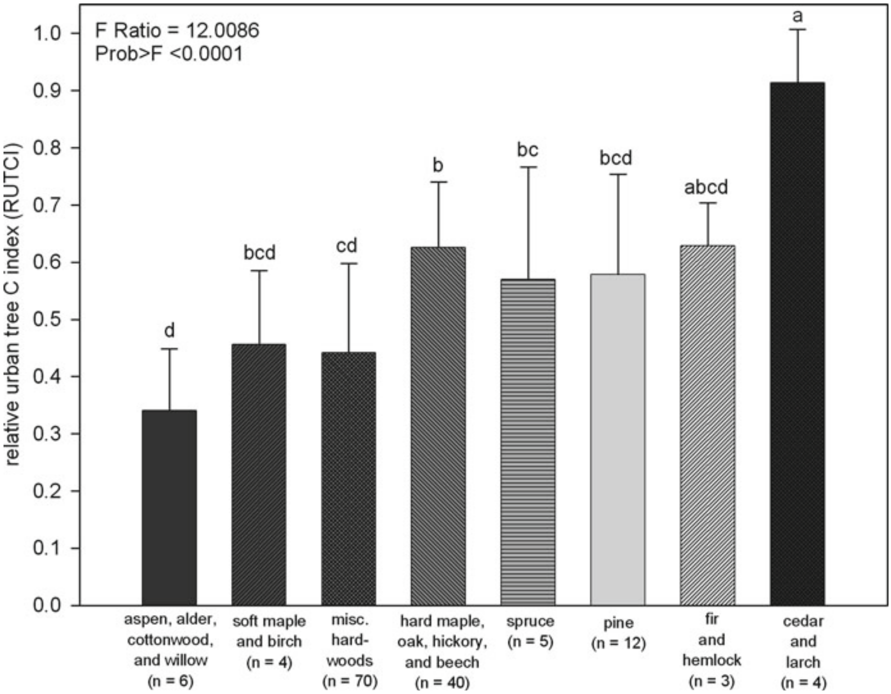

Fig. 6.3 Relative urban tree C index for species for phylogenetic species groups. Standard error of the means are shown and letters indicate mean separations with Tukey's HSD test

tive values for principal component 1, but negative values for principal component 2, relating high C_{st} and LPP, but low urban stress tolerance. The soft maple and birch, miscellaneous hardwoods, hard maple, oak, hickory, and beech species groups tended to fall close to the origin, suggesting moderate scores for these three indices (Fig. 6.2). The aspen, alder, cottonwood, and willow group was negatively related to both components, showing relatively low scores for the three metrics (Fig. 6.2).

The relative urban tree C index (RUTCI) created by summing the average weighted means of RUST, LPP, and C_{st}, ranged from 0 to 1 with a mean value of 0.52 (Appendix). Ninety percent of the species had RUTCI values less than 0.74 and 10% were less than 0.28. The highest RUTCI scores were for: bald cypress, pond cypress, tamarack (*Larix laricina*), northern white cedar, pin oak (*Quercus palustris*), red oak, white oak, shingle oak, bur oak (*Quercus macrocarpa*), and sweetgum (Appendix). The ten species with the lowest RUTCI scores were: jack pine (*Pinus banksiana*), blackhaw viburnum (*Viburnum prunifolium*), American hazelnut (*Corylus Americana*), red mulberry (*Morus rubra*), American mountain ash (*Sorbus Americana*), European hornbeam (*Carpinus betulus*), flowering dogwood (*Cornus florida*), Mountain-laurel (*Kalmia latifolia*), southern magnolia (*Magnolia grandifolia*), and big-tooth aspen (*Populus grandidentata*) (Appendix).

Significant differences were detected for the RUTCI across the assigned species groups (Fig. 6.3). Cedar and larch had greater RUTCI values than aspen, alder, cottonwood, and willow, soft maple and birch, miscellaneous hardwoods, hard maple, oak, hickory, and beech, pine, and spruce (Fig. 6.3). The RUTCI scores for the hard maple, oak, hickory, and beech and pine groups were significantly greater than for the aspen, alder, cottonwood, and willow group (Fig. 6.3). Although differences were detected among these species groups, it is apparent that further consolidation can be performed in this data set for displaying differences in the RUTCI (Fig. 6.3).

Both the PCA and ANOVA analyses suggest that the eight species groups can be lumped into four major groups for assessing potential urban tree C sequestration: (1) cedar and larch, (2) pine, spruce, fir, and hemlock, (3) oak, maple, hickory, beech, birch, and miscellaneous hardwoods, and (4) aspen, alder, cottonwood, and willow (Table 6.1). Storage and sequestration of C was the greatest in cedar and larch (9,690 kg C and 110.5 kg C $year^{-1}$), followed by pine, spruce, fir, and hemlock (4,453 kg C and 82.4 kg C $year^{-1}$), oak, maple, hickory, beech, birch, and miscellaneous hardwoods (1,097 kg C and 36.4 kg C $year^{-1}$), and aspen, alder, cottonwood, and willow (977 kg C and 70.3 kg C $year^{-1}$). Sequestration rate was greater in the aspen, alder, cottonwood, and willow compared to the oak, maple, hickory, beech, birch, and miscellaneous hardwoods, likely as a result of longer lifespans of the latter group. The RUTCI scores were 0.91 for cedar and larch, 0.58 for pine, spruce, fir, and hemlock, 0.51 for oak, maple, hickory, beech, birch, and miscellaneous hardwoods, and 0.34 for aspen, alder, cottonwood, and willow (Table 6.1). Tree lifespan differences followed a similar pattern and was well-correlated ($r = 0.6309$, $p < 0.0001$) with the RUTCI (Table 6.1). Tree size was not correlated to RUTCI and significant differences were not detected among these four species groups (Table 6.1). Tolerance to urban stress (i.e., RUST) and wood density were two functional traits correlated with RUTCI, however the relationships were not as strong as compared to tree lifespan (Table 6.1).

The summary of urban tree populations for four Illinois, U.S.A. cities shows that nine species: green ash (*Fraxinus pennsylvanica*), honeylocust (*Gleditsia triacanthos*), hackberry (*Celtis occentalis*), basswood (*Tilia Americana*), American elm, white ash (*Fraxinus Americana*), silver maple (*Acer saccharinum*), and sugar maple (*Acer saccharum*) comprise 30.3–79.8% of the urban tree populations (Table 6.2). The mean RUTCI of these nine species was 0.57, mean C_{st} was 1,444 kg C, mean RUST score was 1.94, and mean LPP was 471 (Table 6.2). All metrics for these most popular urban tree species suggest that they are intermediate in terms of their C sequestration potential. Consequently, urban forest C storage potential could likely be improved by increasing representation of trees with higher RUTCI scores (e.g., cypress, larch, and oak) (Appendix). Furthermore, these nine most popular urban tree species are all considered targets for recent urban tree epidemics, such as emerald ash borer (*Agrilus planipennis*), Asian long-horned beetle (*Anoplophora glabripennis*), and Dutch elm's disease. If this survey is representative of urban tree populations throughout the U.S.A., the current C storage of urban trees may be in danger.

Table 6.1 Mean values and (standard error) for major species groups and specific gravity, size (maximum diameter at breast height), lifespan, last positive point (LPP), urban stress tolerance (RUST), carbon storage and sequestration, and the relative urban tree carbon index (RUTCI)

Parameter	Cedar and larch	Pine, spruce, fir, and hemlock	Oak, maple, hickory, beech, and misc. hardwoods	Aspen, alder, cottonwood, and willow	ANOVA F Ratio *Prob > F*	Correlation with RUTCI r value *Prob > F*
Specific gravity (dry)	0.44^{b}	0.45^{b}	0.58^{a}	0.38^{b}	21.3309	0.1924
	(0.05)	(0.02)	(0.10)	(0.10)	*<0.0001*	*0.0208*
Size (dbh, cm)	68.8^{a}	86.9^{a}	70.4^{a}	81.2^{a}	1.2812	0.0748
	(8.2)	(15.1)	(2.8)	(8.2)	*0.2833*	*0.3745*
Lifespan (years)	323^{a}	202^{b}	152^{c}	70^{d}	11.6148	0.6309
	(73)	(17)	(7)	(17)	*<0.0001*	*<0.0001*
LPP (years)	765^{a}	$502^{a,b}$	$363^{b,c}$	77^{c}	6.4586	0.7128
	(128)	(67)	(26)	(17)	*0.0004*	*<0.0001*
Urban stress tolerance (RUST)	3.37^{a}	0.38^{b}	$1.97^{a,b}$	$1.18^{a,b}$	9.7081	0.4393
	(0.29)	(0.25)	(0.13)	(0.73)	*<0.0001*	*<0.0001*
C_{st} (kg C)	$9{,}690^{a}$	$4{,}453^{b}$	$1{,}097^{c}$	977^{c}	71.8026	0.5263
	(2,671)	(669)	(53)	(137)	*<0.0001*	*<0.0001*
C_{seq} (kg CO_2 year^{-1})	110.5^{a}	82.4^{a}	36.4^{b}	$70.3^{a,b}$	8.0292	0.1371
	(31.6)	(11.9)	(4.5)	(16.9)	*<0.0001*	*0.1011*
RUTCI	0.91^{a}	0.58^{b}	$0.51^{b,c}$	0.34^{c}	11.2671	1.000
	(0.05)	(0.04)	(0.02)	(0.04)	*<0.0001*	*<0.0001*

Superscript letters show mean separations ($p<0.05$) Tukey's HSD test

Table 6.2 Percent of urban trees in four cities in Illinois, U.S.A. and the associated relative urban stress tolerance (RUST), last positive point (LPP), carbon storage and sequestration, and the relative urban tree carbon index (RUTCI)

Genus species	Chicago, IL (%)	Elburn, IL (%)	Lincoln-wood, IL (%)	Oak Park, IL (%)	RUST	LPP (years)	C_{st} (kg C)	C_{seq} (kg CO_2 year^{-1})	RUTCI
Acer platanoides	4.0	2.4	27.1	13.3	3.17	60	1,837	135	0.57
Fraxinus pennsylvanica	4.9	23.4	15.8	8.4	1.63	480	736	22	0.52
Gleditsia triacanthos	3.3	15.4	15.2	4.4	3.67	480	736	24	0.66
Celtis occidentalis	<1	7.2	<1	8.3	2.27	480	736	18	0.56
Tilia americana	<1	15.5	2.3	11.0	1.09	720	736	15	0.50
Ulmus americana	4.5	<1	1.3	12.0	4.2	720	736	22	0.73
Fraxinus americana	6.0	9.6	3.7	4.8	2.3	720	736	15	0.61
Acer saccharinum	4.6	2.5	3.4	3.9	1.73	60	1,837	108	0.46
Acer saccharum	<1	2.8	3.6	3.5	–0.72	240	1,837	60	0.43
Total or (mean)	**30.3**	**79.8**	**73.4**	**69.6**	**(1.94)**	**(471)**	**(1,444)**	**(43)**	**(0.57)**

6.4 Conclusions

These results suggest that the urban forest C storage potential can be improved by selecting species with traits that maximize C sequestration. Our results suggest that tree lifespan is a key attribute for maximizing C sequestration. Higher tolerances to urban stresses also seem to be important for maximizing C storage potential. Furthermore, trees with higher wood densities appear to be better at C sequestration. Maximum tree size does not appear to be as important of a functional trait for distinguishing an urban tree species' ability to sequester C. These 145 urban tree species were grouped into four classes for assessing C storage potential. Cedar and larch trees ranked highest, followed by pine, spruce, fir and hemlock, then oak, maple, hickory, beech, birch, and miscellaneous hardwoods, and lastly, aspen, alder, cottonwood, and willow. The metrics discussed in this paper provide a much needed framework for assessing relative C storage potential of urban trees. However, many urban areas have severe site limitations, and it is imperative to match the tree's ability to establish and grow with existing site conditions. It appears that the current urban forest C storage pool may be threatened by invasive pests, such as, the Emerald ash borer. More efforts should be directed towards prolonging the C storage potential of these threatened urban trees by adopting methods to increase urban wood utilization and increase C restitution from tree biomass to storage in urban soils. Finally, increasing the species diversity of urban forests will allow these systems to become more resilient and resistant to insect and disease epidemics, thus increasing the C storage potential of urban forests.

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Appendix: Specific Gravity, Size, Lifespan, Last Positive Point (LPP), Relative Urban Stress Tolerance (RUST), Carbon Storage (C_{st}) and Sequestration (C_{seq}), and Relative Urban Tree Carbon Index for 145 Urban Tree Species

Genus species (x cultivar)	Specific gravity (dry)	DBH (cm)	Lifespan (years)	LPP (years)	RUST	C_{st} (kg C)	C_{seq} (kg CO_2 year^{-1})	RUTCI
Abies balsamea	0.35	50	150	240	2.37	4,755	116.1	0.70
Acer buergeranium	0.54	180	150	160	2.18	1,837	44.9	0.56
Acer campestre	0.54	142	150	160	4.14	1,837	44.9	0.69
Acer macrophyllum	0.54	143	15	240	1.11	1,837	448.5	0.51
Acer negundo	0.54	72	63	60	1.40	1,837	107.6	0.43
Acer platanoides	0.54	56	50	60	3.17	1,837	134.6	0.57
Acer rubrum	0.54	56	125	360	1.40	1,837	53.8	0.57
Acer saccharinum	0.47	70	63	60	1.73	1,837	107.6	0.46
Acer saccharum	0.63	70	113	240	–0.72	1,837	59.8	0.43
Aesculus ×carnea	0.43	56	250	960	0.00	736	10.8	0.48
Aesculus glabra	0.43	30	125	160	1.68	736	21.6	0.39
Ailanthus altissima	0.43	66	50	60	3.65	736	53.9	0.45
Alnus glutinosa	0.41	122	50	60	1.21	1,113	81.6	0.34
Alnus serrulata	0.41	73	150	160	2.53	293	7.1	0.36
Amelanchier arborea	0.50	66	150	160	0.72	736	18.0	0.29
Asimina triloba	0.49	46	150	160	1.27	156	3.8	0.24
Betula alleghaniensis	0.62	56	250	720	1.31	1,189	17.4	0.61
Betula nigra	0.61	56	63	60	3.03	1,189	69.7	0.51
Betula papyrifera	0.55	70	68	60	2.95	506	27.5	0.32
Betula populifolia	0.61	40	25	60	1.43	1,189	174.3	0.38
Carpinus betulus	0.61	66	50	60	0.12	736	53.9	0.19
Carpinus caroliniana	0.61	56	125	160	1.82	673	19.7	0.33
Carya aquatica	0.72	70	150	360	3.12	1,837	44.9	0.71
Carya cordiformis	0.72	49	175	240	1.22	1,837	38.4	0.53

(continued)

Appendix (continued)

Genus species (x cultivar)	Specific gravity (dry)	DBH (cm)	Lifespan (years)	LPP (years)	RUST	C_{st} (kg C)	C_{seq} (kg CO_2 year^{-1})	RUTCI
Carya glabra	0.75	56	200	420	−1.42	1,837	33.6	0.47
Carya illinoensis	0.72	56	275	720	1.60	1,837	24.5	0.70
Carya laciniosa	0.72	81	250	720	0.88	1,837	26.9	0.63
Carya ovata	0.72	73	175	240	4.11	1,837	38.4	0.73
Carya tomentosa	0.72	73	200	420	1.72	1,837	33.6	0.64
Catalpa speciosa	0.58	54	63	60	3.65	736	43.2	0.45
Celtis laevigata	0.53	73	150	720	3.53	736	18.0	0.69
Celtis occidentalis	0.53	73	150	480	2.27	736	18.0	0.56
Cephalanthus occidentalis	0.55	73	150	160	2.20	528	12.9	0.34
Cercidiphyllum japonicum	0.52	73	150	480	0.82	736	18.0	0.45
Cercis canadensis	0.66	73	60	60	2.38	736	44.9	0.37
Chionanthus retusus	0.58	73	150	160	2.45	736	18.0	0.45
Chionanthus virginicus	0.58	73	150	160	2.46	452	11.0	0.36
Cornus amomum	0.55	56	150	160	3.45	736	18.0	0.51
Cornus florida	0.55	56	150	160	−1.78	736	18.0	0.22
Cornus sericea	0.55	56	150	160	1.97	736	18.0	0.41
Corylus americana	0.55	49	150	160	0.21	90	2.2	0.17
Crataegus viridis	0.50	56	50	60	2.39	404	29.6	0.28
Diospyros virginiana	0.55	56	63	420	3.05	736	43.2	0.60
Fagus grandifolia	0.64	73	250	720	2.05	1,837	26.9	0.74
Fagus sylvatica	0.64	73	250	420	−0.74	1,837	26.9	0.48
Fraxinus americana	0.60	56	175	720	2.30	736	15.4	0.61
Fraxinus pennsylvanica	0.56	46	125	480	1.63	736	21.6	0.52
Ginkgo biloba	0.55	56	250	720	3.45	736	10.8	0.69
Gleditsia triacanthos inmis	0.66	56	113	480	3.67	736	24.0	0.66
Gymnocladus dioicus	0.66	56	113	720	4.20	736	24.0	0.73

(continued)

Appendix (continued)

Genus species (x cultivar)	Specific gravity (dry)	DBH (cm)	Lifespan (years)	LPP (years)	RUST	C_{st} (kg C)	C_{seq} (kg CO_2 year^{-1})	RUTCI
Hamamelis virginiana	0.52	24	150	160	1.46	736	18.0	0.37
Ilex opaca	0.55	44	150	160	5.40	736	18.0	0.56
Ilex verticillata	0.55	56	150	160	3.00	736	18.0	0.48
Juglans nigra	0.55	73	125	960	2.68	736	21.6	0.69
Juniperus virginiana	0.47	73	250	420	3.10	736	10.8	0.60
Kalmia latifolia	0.55	56	150	160	0.88	452	11.0	0.23
Koelreuteria paniculata	0.43	56	150	160	3.75	736	18.0	0.53
Larix laricina	0.52	73	165	720	2.66	2,417	53.7	0.84
Lindera benzoin	0.55	56	150	160	2.84	20	0.5	0.37
Liquidambar styraciflua	0.52	56	225	960	3.87	736	12.0	0.76
Liriodendron tulipifera	0.49	56	150	720	0.71	736	18.0	0.47
Maackia amurensis	0.66	56	150	160	1.95	736	18.0	0.41
Magnolia acuminata	0.48	56	100	720	1.37	736	27.0	0.53
Magnolia grandifolia	0.50	56	50	60	0.78	736	53.9	0.23
Magnolia virginiana	0.49	56	150	160	4.95	736	18.0	0.56
Malus spp.	0.50	56	50	60	1.38	736	53.9	0.28
Morus rubra	0.50	46	50	60	−0.67	736	53.9	0.17
Nyssa sylvatica	0.55	56	135	360	2.88	736	20.0	0.55
Ostrya carpinifolia	0.55	116	150	160	1.27	736	18.0	0.34
Ostrya virginiana	0.55	13	125	160	3.65	736	21.6	0.52
Oxydendrum arboreum	0.55	56	150	160	1.88	736	18.0	0.40
Picea abies	0.39	56	250	420	1.52	6,058	88.8	0.70
Picea glauca	0.36	56	275	420	−0.67	2,426	32.3	0.55
Picea mariana	0.42	56	163	160	1.51	575	13.0	0.29
Picea pungens	0.39	56	288	420	0.73	6,439	82.0	0.62
Picea rubens	0.40	56	250	720	1.32	3,826	56.1	0.74
Pinus banksiana	0.43	56	35	60	−0.61	195	20.4	0.08
Pinus echinata	0.51	56	150	480	0.21	1,918	46.8	0.60
Pinus elliottii	0.59	56	150	720	0.11	3,097	75.6	0.64
Pinus mugo	0.47	250	150	160	2.15	9,831	240.1	0.65
Pinus nigra	0.47	250	150	360	0.31	2,585	63.1	0.55
Pinus palustris	0.59	207	150	720	0.21	3,516	85.9	0.66

(continued)

Appendix (continued)

Genus species (x cultivar)	Specific gravity (dry)	DBH (cm)	Lifespan (years)	LPP (years)	RUST	C_{st} (kg C)	C_{seq} (kg CO_2 year^{-1})	RUTCI
Pinus ponderosa	0.40	56	325	960	0.65	9,831	110.8	0.74
Pinus resinosa	0.46	56	325	960	–0.11	3,303	37.2	0.68
Pinus strobus	0.35	56	275	960	–1.39	9,831	131.0	0.68
Pinus sylvestris	0.47	56	150	360	0.31	5,519	134.8	0.56
Pinus taeda	0.51	140	150	480	1.51	5,614	137.1	0.72
Pinus virginiana	0.48	56	50	60	0.31	1,056	77.4	0.28
Platanus × acerifolia	0.49	56	250	960	2.97	736	10.8	0.70
Platanus occidentalis	0.49	56	325	960	2.48	736	8.3	0.67
Poplulus tremuloides	0.36	73	55	60	–0.23	1,113	74.1	0.26
Populus deltoides	0.40	73	68	60	3.82	1,113	60.4	0.54
Populus grandidentata	0.36	73	65	60	–1.01	1,113	62.7	0.24
Prunus pensylvanica	0.50	56	50	60	2.65	528	38.7	0.31
Prunus serotina	0.50	56	150	480	1.90	736	18.0	0.54
Prunus virginiana	0.50	56	150	160	0.20	736	18.0	0.27
Pseudotsuga menziesii	0.48	56	250	960	–1.25	1,982	29.0	0.64
Quercus alba	0.68	73	375	720	2.90	1,837	17.9	0.79
Quercus bicolor	0.72	73	150	360	3.87	1,837	44.9	0.75
Quercus chrysolepsis	0.88	73	250	480	0.81	1,837	26.9	0.58
Quercus coccinea	0.67	73	250	960	1.63	1,837	26.9	0.75
Quercus douglasii	0.69	73	250	420	0.81	1,837	26.9	0.55
Quercus imbricaria	0.69	73	225	420	3.95	1,837	29.9	0.79
Quercus kelloggii	0.61	73	250	720	0.81	1,837	26.9	0.62
Quercus laurifolia	0.63	73	50	60	3.02	1,837	134.6	0.56
Quercus lobata	0.69	73	250	420	0.81	1,837	26.9	0.55
Quercus lyrata	0.63	73	250	480	1.53	1,837	26.9	0.65
Quercus macrocarpa	0.64	73	250	720	2.50	1,837	26.9	0.77

(continued)

Appendix (continued)

Genus species (x cultivar)	Specific gravity (dry)	DBH (cm)	Lifespan (years)	LPP (years)	RUST	C_{st} (kg C)	C_{seq} (kg CO_2 year^{-1})	RUTCI
Quercus michauxii	0.67	73	150	360	2.60	1,837	44.9	0.67
Quercus muehlenbergii	0.69	73	175	420	1.05	1,837	38.4	0.57
Quercus nigra	0.63	73	50	60	1.21	1,837	134.6	0.40
Quercus nuttallii	0.69	73	150	360	3.86	1,837	44.9	0.75
Quercus palustris	0.63	73	150	720	3.35	1,837	44.9	0.82
Quercus phellos	0.69	73	250	420	1.93	1,837	26.9	0.65
Quercus prinus	0.66	73	250	720	0.62	1,837	26.9	0.60
Quercus robur	0.69	73	250	420	0.24	1,837	26.9	0.52
Quercus rubra	0.63	29	250	960	2.32	1,837	26.9	0.80
Quercus shumardii	0.69	56	150	360	3.57	1,837	44.9	0.73
Quercus velutina	0.61	70	175	360	1.65	1,837	38.4	0.60
Quercus virginiana	0.88	56	250	960	1.11	1,837	26.9	0.69
Rhus glabra	0.69	56	13	60	2.73	194	56.9	0.30
Robinia pseudoacacia	0.69	56	63	60	1.93	736	43.2	0.34
Salix nigra	0.39	73	30	60	0.75	1,113	135.9	0.30
Sambucus canadensis	0.55	56	150	160	1.56	736	18.0	0.38
Sassafras albidum	0.55	56	63	60	3.30	736	43.2	0.43
Sorbus americana	0.50	56	50	60	–0.17	736	53.9	0.18
Syringa reticulata	0.58	56	150	160	1.90	736	18.0	0.41
Taxodium ascendens	0.46	73	250	960	3.78	13,699	200.7	0.99
Taxodium distichum	0.46	73	500	960	3.88	13,699	100.4	1.00
Thuja occidentalis	0.31	56	375	420	3.15	8,944	87.4	0.83
Tilia americana	0.37	151	175	720	1.09	736	15.4	0.50
Tilia cordata	0.37	102	150	720	–1.14	736	18.0	0.41
Tilia euchlora	0.37	55	150	480	–0.26	736	18.0	0.39
Tilia tomentosa	0.37	155	150	240	0.97	736	18.0	0.37
Tsuga canadensis	0.40	56	350	420	–1.63	6,690	70.0	0.55

(continued)

Appendix (continued)

Genus species (x cultivar)	Specific gravity (dry)	DBH (cm)	Lifespan (years)	LPP (years)	RUST	C_{st} (kg C)	C_{seq} (kg CO_2 year^{-1})	RUTCI
Ulmus americana	0.50	124	125	720	4.20	736	21.6	0.73
Ulmus parvifolia	0.55	36	150	360	2.31	736	18.0	0.50
Ulmus pumila	0.55	92	150	720	1.41	736	18.0	0.54
Ulmus rubra	0.55	112	150	360	4.98	736	18.0	0.63
Ulmus serotina	0.55	180	150	480	1.21	736	18.0	0.47
Ulmus thomasii	0.63	104	150	240	1.31	736	18.0	0.39
Viburnum prunifolium	0.55	118	50	60	–0.75	556	40.7	0.09
Viburnum trilobum	0.55	180	150	160	2.55	736	18.0	0.46
Zelkova serrata	0.50	115	150	240	2.30	736	18.0	0.48
Mean	**0.55**	**73**	**160**	**382**	**1.75**	**1,797**	**46.3**	**0.52**
Median	**0.55**	**56**	**150**	**360**	**1.67**	**736**	**26.9**	**0.54**
10th percentile	**0.40**	**55**	**50**	**60**	**–0.21**	**692**	**15.4**	**0.28**
90th percentile	**0.69**	**116**	**250**	**720**	**3.74**	**2,994**	**98.0**	**0.74**

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Chapter 7
Carbon Storage in Some Urban Forest Soils of Columbus, Ohio, USA

Klaus Lorenz and Rattan Lal

Abstract Soils store more carbon (C) in soil organic matter (SOM) and in carbonates than the vegetation C and atmospheric carbon dioxide (CO_2)-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 = 10^6 g) C ha^{-1} 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 notrgen (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*. 0.7–0.9% 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 C ha^{-1} 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 (97–145 Mg C ha^{-1}). 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_2) 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_2 by taking up between 1.58 and 2.37 petagram (1 Pg = 10^{15} g) C $year^{-1}$ (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_2) 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_2 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_2 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_2 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_2 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_2 in urban ecosystems as urban trees fix CO_2 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_2 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 C ha^{-1} (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 $year^{-1}$ (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 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 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 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 forest-land 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 or 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^{-1}. 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^{-1}. 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^{-1} (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^{-1} and store 157.1 Mg C ha^{-1} 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^2, 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.colscompact.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 (ρ_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 (Mg ha^{-1}) for a specific layer of thickness d (m) was calculated using Eq. 7.1 below (Lorenz and Lal 2007):

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

where ρ_b is the bulk density (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)

		Nitrogen (%)			Carbon (%)		
Site	Depth (cm)	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 were 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 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^{-1}), and higher in 10–20 cm compared to that in 30–40 cm depth (24.0 Mg C ha^{-1} *vs.* 16.8 Mg C ha^{-1}, respectively). Differences in soil C pool were rather small among other depths. In total, 210.9 Mg C ha^{-1} 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^{-1}, 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^{-1} 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

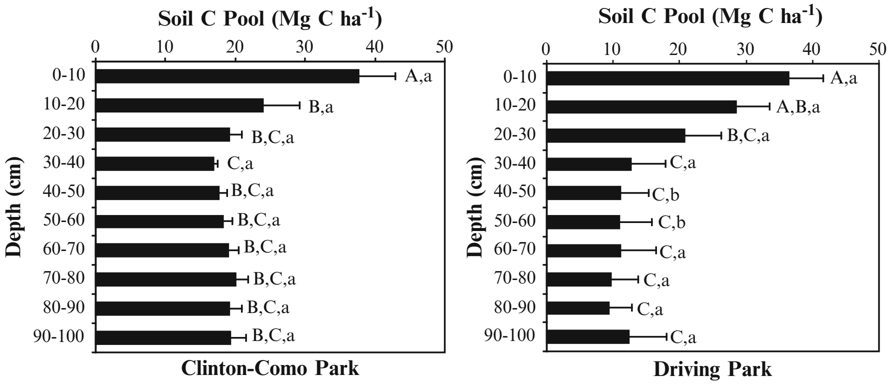

Fig. 7.1 Soil carbon pool (Mg C ha^{-1}) to 1-m depth at two urban forests in Columbus ($N=10$; ±SD; means for each site not sharing a common capital letter are statistically different among depths; means for each depth not sharing a common lowercase letter are statistically different among sites, ANOVA, 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^{-1} to 1-m depth, and up to ~1,150 Mg SOC ha^{-1} may be stored in Histosols covered by forest (Tan et al. 2004). Further, urban forests in Atlanta store 77 Mg SOC ha^{-1} and those in Baltimore 115.6 Mg SOC ha^{-1} 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_2 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_2. 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_2 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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Part III
Turfgrass and Home Lawns

Chapter 8
Carbon Dynamics and Sequestration in Urban Turfgrass Ecosystems

Yaling Qian and Ronald Follett

Abstract Urbanization is a global trend. Turfgrass covers 1.9% of land in the continental US, occupying about 16 million ha. In this article, we review existing literature associated with carbon (C) pools, sequestration, and nitrous oxide emission of urban turfgrass ecosystems. Turfgrasses exhibit significant carbon sequestration (0.34–1.4 Mg ha^{-1} year^{-1}) during the first 25–30 years after turf establishment. Several studies have reported that residential turfgrass soil can store up to twofold higher soil organic carbon (SOC) content than agricultural soils. Published research suggests that the dynamics of nitrogen (N) is controlled by C transformation. Turfgrass areas have high levels of SOC and microbial biomass creating a carbon-based "sink" for inorganic N. Therefore, lower than "expected" nitrate leaching and N_2O emissions have been measured in the majority of the experiments carried out for turfgrass ecosystems. Increased SOC in turfgrass soil can result from: (1) returning and recycling clippings, (2) appropriate and efficient-fertilizer use, and (3) irrigation based on turfgrass needs. Some turfgrass management practices (such as fertilization, mowing, and irrigation) carry a carbon "cost". Therefore turfgrass's contribution to a sink for carbon in soils must be discounted by fuel and energy expenses and fertilizer uses in maintaining turf, and the flux of N_2O. More work is needed to evaluate the carbon sequestration, total carbon budget, and fluxes of the other greenhouse gases in turfgrass systems.

Keywords Soil carbon sequestration • Turfgrass • Irrigation • Fertilization • Clipping

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List of Abbreviations

C	carbon
N	nitrogen
SOC	soil organic carbon
SON	soil organic nitrogen
SOM	soil organic matter

8.1 Introduction

Urbanization is a global trend. Large areas of native vegetation, agricultural land, and forest areas have been converted into urbanized landscapes. Accompanying this growth and development is a rapid increase in the area of turfgrass, such as home lawns, commercial landscapes, parks, golf courses, recreational facilities, and other greenbelts. Turfgrass ecosystems provide excellent soil erosion control, dust stabilization, flood control, and urban heat dissipation (Beard and Green 1994). Partly because turf provides aesthetic benefits and produces an ideal surface for leisure activities, no city or suburban community is devoid of turfgrass. Milesi et al. (2005) estimated that among the total land in the United States devoted to urban development, 39–54% is covered by turfgrass. It is estimated that turfgrass covers 1.9% of land in the continental US, occupying about 16 million ha (Mha).

These urban turfgrass systems represent grasslands that experience disturbances. The disturbance factors include initial construction and movement of soil; establishment of improved turfgrass species; subsequent water and N addition; and continuous mowing and traffic. Irrigation and fertilization are two of the major management components in maintaining aesthetically appealing turfgrass in urban and suburban landscapes. These human dominated influences may alter the storage and fluxes of C and N, with possible influence on greenhouse gas (GHG) budget in urban and suburban areas. Research information is needed in regard to C dynamics and sequestration capability of turfgrasses under different management regimes. In this chapter, soil C sequestration is defined as the ecosystem's ability of capturing and long-term storage of C as soil organic matter or SOM (Lal and Follett 2009).

8.2 Carbon Dynamics and Sequestration in Turfgrass-Soil Systems

Soil organic C (SOC) dynamics are complex. In general, SOC comprises about 58% of the SOM mass, which has profound influence on ecosystem sustainability, soil fertility, and soil structure (Tabatabai 1996). It is a major repository and reserve of plant nutrients, especially nitrogen (N), phosphorus (P), sulfur (S), and potassium (K).

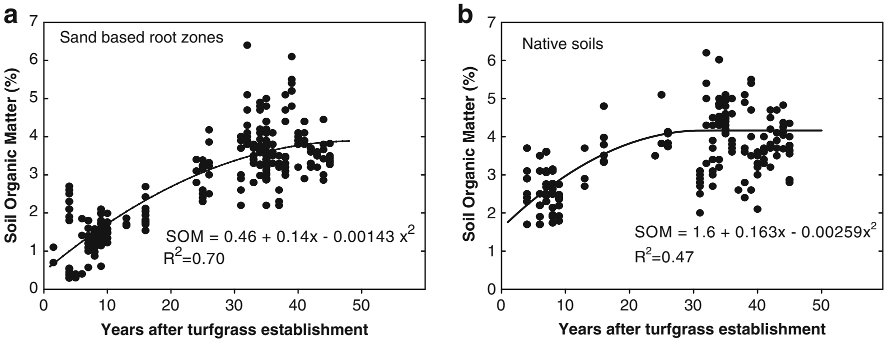

Fig. 8.1 Changes in soil organic matter with time since turf establishment in (**a**) sand based root zone and (**b**) native soil

Changes in SOM also influence the air-filled porosity, water retention, and other soil health parameters. Fresh organic matter, such as plant material, is partitioned into either structural or metabolic SOM pools based on the lignin: N ratio. The SOM is partitioned into three pools: active, slow, and passive SOC pools (Parton et al. 1987). The active pool consists of microbes and microbial byproducts, having the most rapid turnover rate. The slow pool represents stabilized decomposition products with an intermediate turnover rate; and the passive pool is recalcitrant SOC with a turnover rate of hundreds or thousands of years. The turnover times of these pools are functions of the soil's temperature, aeration, water availability, and nutrient status. Thus, soil texture, temperature, oxygen, nutrient and water availability, plant productivity and above and belowground biomass characteristics (C:N ratio, lignin content, etc.) are all important variables that influence SOC changes.

Due to irrigation and fertilization input, turfgrasses have high root and shoot biomass productivity (Falk 1976; 1980), and high C input into the soil as roots, litters, and exudates. Moreover, it is common for turfgrass to be established on top of subsoil, which displays low SOC after urban development and construction disturbance. The initial low SOC, high productivity, and perennial nature of turfgrass, and minimal tillage all contribute to the SOC sequestration potential of turfgrass.

A previous study used long-term golf course soil testing data in Colorado to determine SOC sequestration rates (Qian and Follett 2002). A total of ~690 data sets indicate a strong pattern of SOM response to decades of turfgrass culture (Fig. 8.1). Assuming that SOC comprises 58% of SOM mass and using measured soil bulk density data (1.6 $g\,cm^{-3}$ for putting greens and 1.5 $g\,cm^{-3}$ for fairways), the SOM increases presented in Fig. 8.1 would transfer to total SOC sequestration rates of 0.9 and 1.0 Mg ha^{-1} $year^{-1}$ during the first 25–30 years after turfgrass establishment in fairways and putting greens, respectively. In this study, soils were only sampled to 11.4 cm depth. However, if deep C-sequestration does occur, these estimates would be conservative and the estimated period of potential sequestration would be longer. These rates are comparable to or exceed those reported by Follett et al. (2001) for US land that has been converted to the conservation reserve program (CRP).

Qian et al. (2010) also reported SOC changes, soil C sequestration, and SOC decomposition rates along a golf course management gradient using C isotopic techniques. It was found that different carbon sequestration rates were observed under different management regimes, ranging from 0.34 to 0.78 Mg ha^{-1} $year^{-1}$ during the 4-years-study period. The SOC sequestration rates were 0.74, and 0.78 Mg ha^{-1} $year^{-1}$ for irrigated-fine fescue *(Festuca spp.)* and creeping bentgrass (*Agrostis palustris* Huds.), respectively, which are higher than those of non-irrigated fine fescue and irrigated Kentucky bluegrass *(Poa pratensis* L.*)*. This range of SOC sequestration is in agreement with the following reported studies. Bruce et al. (1999) estimated a gain of 0.6 Mg C ha^{-1} $year^{-1}$ for previously cultivated lands that had been reseeded to grass. Post and Kwon (2000) compiled data for soil C in areas where grasslands have been allowed to develop on previously disturbed lands and reported that the average rates of C accumulation during the early grassland establishment were 0.33 Mg ha^{-1} $year^{-1}$. Many studies indicated that the ability of turfgrass for SOC sequestration in soils is generally in the top layer and decreases strongly with depth. The C isotope technique (Qian et al. 2010) also showed that 4 years after turfgrass establishment on a previous corn (*Zea mays* L.)-soybean (*Glycine max* L.) field in north central USA, about 17–24% of SOC was derived from turfgrass at 0–10 cm depth. At 10–20 cm depth, there were striking differences among turfgrass species. For shallow rooted Kentucky bluegrass and creeping bentgrass, only about 1% and 4% of SOC was derived from turfgrass at 10–20 cm, whereas for the deep-rooted fine fescue about 10–13% was derived from turfgrass. Deep-rooted grasses can sequester SOC deep in the soil profile. When the data were combined for 2 depths, about 10–18% of the SOC pool was derived from the turfgrass.

Also in the Front Range of Colorado, Golubiewski (2006) reported that turfgrass sites had higher biomass productivity and larger SOC pools, more than double in some cases, than native shortgrass steppe or agricultural ecosystems. While native grasslands store about 90% of C belowground, urban landscapes store a decreasing proportion of the total C belowground through time, reaching about 70% after 30–40 years following the construction of landscapes. Both in Colorado and Arizona, residential turfgrass had about twofold higher SOC density than native ecosystems, likely due to the irrigation input in these semi-arid and arid climates (Pouyat et al. 2009; Zhu et al. 2006). Pouyat et al. (2009) studied SOC of turfgrass systems in Baltimore and Denver metropolitans in comparison with their native ecosystems (hardwood deciduous forest and shortgrass steppe, respectively). They reported that in both areas, residential turfgrass soil had up to twofold higher SOC concentration than did soils under native ecosystems. The regional differences in SOC storage between Denver and Baltimore were much smaller in turfgrass systems than in native ecosystems.

Fossil fuel use is directly and indirectly associated with turfgrass management practices such as mowing, fertilizing, and irrigation. To determine the impact of turfgrass on the net CO_2 (atmosphere's GHG) budget, fuel expenses in maintaining turfgrass, fertilizer uses, energy for pumping water to irrigate, and the fluxes of other GHGs (mainly N_2O) must also be considered in addition to SOC sequestration. In California, where the growing season is much longer than in many other US regions, the rates of SOC sequestration of turfgrass, and CO_2 emissions associated with fuel and energy consumption, fertilizer C cost, and the fluxes of N_2O were estimated in a park (Townsend-Small and Czimczik 2010). It was observed that rate of SOC sequestration of turfgrass was 1.4 Mg ha^{-1} year^{-1} in the top 20 cm depth. The authors suggested that this rate of SOC sequestration is close to that observed in re-growing forests in the north-eastern USA. Unfortunately, the authors miscalculated the CO_2 emission by fuel use for managing turf, which led to authors' conclusion, i.e. turfgrass management generated more CO_2 than what was sequestered. With the error corrected, the SOC sequestration rate on the study site (park) would be at least equal or exceed the C cost of managing the turf (Toro Company 2010).

As discussed above, the reported SOC sequestration rate ranged from 0.34 to 1.4 Mg C ha^{-1} year^{-1}. The variations of reported SOC sequestration rate in turfgrass soils could be attributed to many factors, including but not limited to duration of the growing season, depth of soil sampled, variability in weather conditions, soil type, turfgrass types, and management regimes.

8.3 Soil Organic Carbon and Soil Organic Nitrogen Relations

Both SOC and N cycles are interacted and coupled. Coupling C and N can allow a better understanding of both soil C and N dynamics. The following is a hypothetical description of SOC and N relationship for land that is converted to turfgrass systems from agricultural use or native grass steppe in Colorado.

We hypothesize that a rapid C sequestration in relatively young turf sites (< 25–30 years) favors N immobilization (Diagram 8.1). At this stage, system N gain appreciably exceeds system N loss and N input through fertilization is retained in the system. Therefore, vadose-zone nitrate level (therefore, nitrate leaching) and N trace gas fluxes in young turf sites may be at minimal levels. However, turf systems have high C turnover. Falk (1976) reported that the root turnover time is 2.4 year for a Kentucky bluegrass lawn at 0–7.5 cm depth, whereas root turnover time for native tall grass prairie is 3.85 year (Dahlman and Kucera 1965). Due to the low biomass C:N ratio, low lignin content, and fast turnover of above and below ground biomass, turf systems reach a relative C equilibrium faster than forest and native grassland. As turf ages, the rate of SOC sequestration decreases, which is consistent with the observation of Qian and Follett (2002) (Fig. 8.1).

In support of the above hypothesis, Porter et al. (1980) reported that total soil N increased as a function of the turf system age and leveled off after ~10–12 years. Moreover, the majority of studies about nitrate leaching have shown that turfgrass presents little risk to the environment from nitrate leaching on newly established turf plots (<10 years) (Cohen et al. 1999; Gold and Groffman 1993; Gross et al. 1990; Miltner et al. 1996; Starr and DeRoo 1981; Petrovic 1990). However, most of these studies were conducted on turf plots less than 10 years old. It is possible to have more N losses in matured turf ecosystems if high rates of N applications continue. For example, a study was conducted by Jiang et al. (2000) on a long-established

Land use conversion to urban turf grassland

⇓

Increased C fixation

Increased production

Increased root C exudation

Increased litter production

Carbon sequestration

⇓

Root N uptake

N immobilization in SOM

Substantial N is retained in the ecosystem as soil organic nitrogen (SON)

⇓

Mineral N pool decrease

⇓

Gaseous losses decrease

N leaching decreasve

⇓

As ecosystems mature, system shift to a new N enriched state.

⇓ *Continuous N addition*

C and N saturation

⇓ *Continuous N addition*

Gaseous losses may increase

N leaching increase

Diagram 8.1 Carbon sequestration and N retention as function to turf ecosystem standing age

turf stand (~ 15 years) where they found that annual leaching of nitrate from a turf system reached about 50 kg N ha^{-1} year^{-1} when 170 kg N ha^{-1} year^{-1} was applied to turf.

Compared with studies of N leaching, few studies have measured trace gas fluxes as driven by turfgrass systems. The nitrogenous trace gases are generated from microbial processes with reactive N in the soil. For example, N_2O is such a trace gas that is also a GHG with 298 times greater global warming potential than CO_2 on per molecular basis and on a 100-year time frame (IPCC 2007). In the nitrification/denitrification processes, N_2O is produced either as a by-product or a reaction intermediate that can diffuse from the microbial reaction site into the soil and eventually move into the atmosphere (Bremner and Blackmer 1981; Firestone and Davidson 1989). Townsend-Small and Czimczik (2010) observed that the annual N_2O emission from turfgrass systems ranged from 1 kg N ha^{-1} year^{-1} when turfgrass annual fertilization rate was 100 to 3 kg N ha^{-1} year^{-1} when turfgrass annual fertilization rate was 750 kg N ha^{-1} year^{-1}. A N fertilization rate of 750 kg N ha^{-1} year^{-1} is an extremely high rate, rarely used in the turfgrass industry. Annual emission of N_2O was measured at 2.4 kg N ha^{-1} year^{-1} in northern Colorado residential lawns when 120 kg N ha^{-1} year^{-1} was applied between June and October (Kaye et al. 2004). Bremer (2006) reported that accumulative emissions of N_2O-N were

1.65 kg ha^{-1} year^{-1} for turfgrass when fertilized with 250 kg ha^{-1} year^{-1} urea-N, 1.60 ha^{-1} year^{-1} when fertilized with 250 kg ha^{-1} year^{-1} ammonium sulfate-N, and 1.01 kg ha^{-1} year^{-1} when fertilized with 50 kg ha^{-1} year^{-1} urea-N. Most of the N_2O emission measurements suggested that N_2O emission from turfgrass systems were lower than agricultural systems and were not as high as expected. Although turf is generally not planted as a harvesting crop, yet it receives irrigation and fertilization. Thus, some researchers expect turf systems to have high N off-fluxes, including leaching and N trace gas fluxes. The current IPCC (Intergovernmental Panel on Climate Change) national inventory methodology for estimating N_2O emissions from soils relates N_2O emissions to fertilizer input. Currently, IPCC estimates that loss of N_2O accounts for 2% of nitrogen added as fertilizer (IPCC 2007).

Dynamics of N is controlled by C transformation. The hypothesis depicted in Fig. 8.1 is further supported by a study conducted in Baltimore (Groffman et al. 2009). In this study, soils were sampled in 14 forest, 10 row crop agriculture, and 10 grass sites. Both SOM and soil nitrate were compared among three ecosystems, and results indicated that forest and urban turfgrass sites exhibited higher SOM, microbial biomass C, and microbial respiration than agricultural sites (Groffman et al. 2009). Urban grasslands had lower levels of nitrate leaching and N_2O emission than row crop sites, but higher levels of nitrate leaching and similar levels of N_2O emission as compared to forest plots. The authors suggested that the surprisingly high N retention in urban turfgrass is driven by high rates of C cycling by soil and vegetation. The cycling of C drives N retention by plant uptake and microbial immobilization, which is tightly linked to SOM dynamics and the C:N ratio of SOM. Turfgrass supported high levels of nitrification, but low levels of soil nitrate because levels of SOM and microbial biomass are high under turfgrass, creating a C-based "sink" for inorganic N, therefore lower than "expected" nitrate leaching and N_2O emissions have been measured in majority of the experiments carried out in the turfgrass ecosystems.

8.4 Management Practices to Increase Carbon Sequestration in Turfgrass Systems

Management of turfgrasses is highly variable, in part because of the different uses, species, nutrient management, clipping management, and input levels. Management practices in turfgrass can play an important role in SOC and nitrogen cycling. The intensity of management influences the soil physical structure as well as moisture retention and distribution. The microbial processes of mineralization (conversion of complex organic molecules to ammonium), nitrification (oxidation of ammonium to nitrite and nitrate), and denitrification (reduction of nitrate to dinitrogen) are regulated by the availability of oxygen, water, and available C for denitrification. Golubiewski (2006) observed that management level dominates the response of turfgrass production and tissue N concentration, which, in turn, influences the amount of C and N stored in the turf site.

Research to document the effects of different management scenarios on SOC and N changes can help in a better understanding of the impact of turfgrass on urban ecosystem C budgets.

8.4.1 Clipping and Fertilization Management on C and N Dynamics

Turfgrass produces a vast amount of clippings every year (Harivandi et al. 2001; Kopp and Guillard 2002), often times being removed because clippings can be unsightly. However, with increasing restrictions of yard waste from landfills and efforts in reducing resource input in turfgrass systems, recycling clippings to turfgrass with a mulching mower has become a more common practice. The N and C interrelationships of clipping management associated with other components (ecosystem age and fertilization) are complex; single period static analysis cannot capture the inter-temporal interactions and carryover effects. One means of studying potential impacts of alternative management systems is computer modeling. To assess clipping management on SOC sequestration potential, the CENTURY model simulation was conducted by Qian et al. (2003).

The CENTURY is an ecosystem model primarily developed to evaluate C, N, S, and P dynamics in the Great Plains Grasslands, based on different C pools namely active, slow and passive SOC with different potential decomposition rates (Parton et al. 1987). The CENTURY model was previously validated for turfgrass ecosystems in Colorado by Bandaranayake et al. (2003). Bandaranayake and colleagues evaluated the CENTURY's performance by comparing long term SOC data from golf courses with age ranging from 1 to 45 years to model-predicted SOC. It was found that the CENTURY's predictions of SOC compared reasonably well with the measured SOC, with regression coefficients of 0.67 for fairways and 0.87 for putting greens (Fig. 8.2).

Qian et al. (2003) tested the long-term (50–100 years) impacts of Kentucky bluegrass clipping management and N fertilization rates on SOC and SON concentrations using the CENTURY model in northern Colorado climates. Four representative management scenarios: the factorial combination of clipping management (removed vs. returned) and two N fertilization levels (150 vs. 75 kg N ha^{-1} $year^{-1}$) were simulated. The simulated results indicated that SOC increased with all management scenarios except that for low N fertilization with clippings removal, in which case SOC did not increase after land use being changed from native grassland to turf (Fig. 8.3). Compared to clippings removed scenario, returning clippings for 10–50 years would increase SOC sequestration by 11–25% under high (150 kg N ha^{-1} $year^{-1}$) N fertilization regime and by 11–59% under low (75 kg N ha^{-1} $year^{-1}$) N fertilization regime. Returning clippings increases SOC sequestration capacity by 14–21 Mg ha^{-1} 50 years after establishment of turf. The simulation showed that clipping management is an important variable affecting SOC concentration. In comparison with clipping removal management, SOC and SON accumulates to greater extents under the clipping recycling treatment despite much lower amount of applied N.

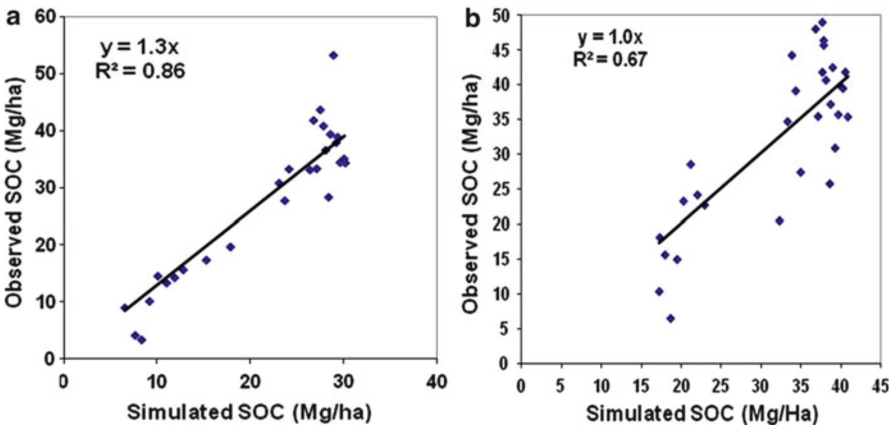

Fig. 8.2 Comparison of measured and simulated soil organic carbon (SOC) in fairways (**a**) and putting greens (**b**) in golf courses near Denver and Fort Collins

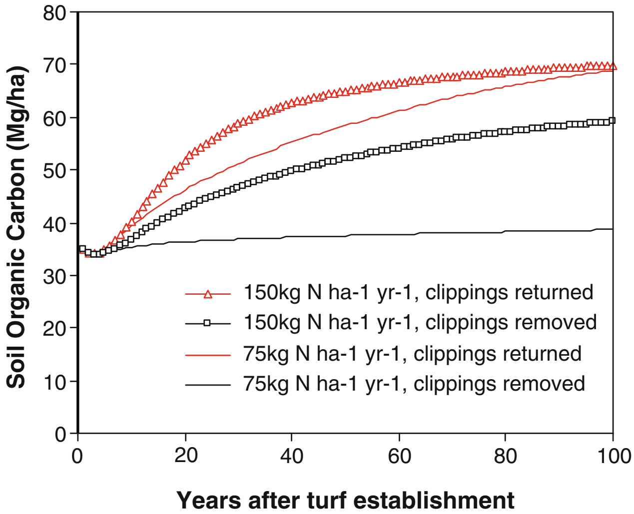

Fig. 8.3 The CENTURY-simulated long-term effects of clipping management and nitrogen fertilization rate on soil organic carbon content. The simulation was done for a residential turfgrass with a loam soil

Availability of N is important in the ecosystem's above- and below ground productivity. A low N fertilization coupled with clipping removal would result in a very low biomass yield and no SOC increase after land use being changed to turf (Fig. 8.3). Simulated SOC dynamics showed that high N availability leads to more rapid increases in SOC after turf establishment and faster arrival at a relatively steady state than low N management (Qian et al. 2003).

The CENTURY model is also used as a management supporting system to generate optimal N fertilization rates as a function of turfgrass age, and with an aim to achieve adequate production under the constraint of minimal N out-flux (Qian et al. 2003). The CENTURY model predicted that, as the age of turf ecosystem increases, N application rate needs to be reduced due to the greater amounts of N mineralized and recycled from SOM. For a site with predominantly a loam soil under clipping returned scenario in northern Colorado, the CENTURY model predicted that optimal productivity and minimum nitrate leaching (< 2 kg N ha^{-1}) could be obtained for an annual N fertilization rate of 150, 100, and 75 kg N ha^{-1} during 1–10, 11–25, and 26–50 years after turfgrass establishment, respectively. In contrast, under clipping removed scenario, the CENTURY model predicted that, to achieve a comparable productivity and turf quality, N fertilization at 200, 150, and 140 kg N ha^{-1} per year would be required for the periods of 1–10, 11–25, 25–50 years after turfgrass establishment, respectively. Returning grass clippings back to turf/soil ecosystem can significantly reduce the fertilization requirements by 25% between 1 and 10 years after turf establishment, 33% between 11 and 25 years after establishment, and 50% between 25 and 50 years after establishment. This reduction in fertilizer levels with clipping return could result in a substantial reduction in N_2O emission. Bremer (2006) suggested that higher N fertilization rate results in greater N_2O emissions from turfgrass systems.

8.4.2 *Other Factors*

In addition to clipping and fertilization management, researchers have found that adequate irrigation and plant growth regulator (trinexapac-ethyl and paclobutrazol) applications would increase SOC input (Qian et al. 2010; Lopez-Bellido et al. 2010). Qian et al. (2010) reported that irrigated-fine fescue had a 42% higher SOC sequestration rate than that of non-irrigated fine fescue. Irrigation increased turfgrass C input to the 0–20 cm soil layer (by 141%) as well as SOC decomposition (by 200%). Blanco-Canqui et al. (2010) reported that SOC concentration and wet aggregate stability increased with an increase in irrigation amount. Lopez-Bellido et al. (2010) reported an increase in SOC when creeping bentgrass fairways were treated with plant growth regulators. Despite the fact that N fertilizer and some agricultural chemicals could increase SOC concentration, these products carry a hidden C “cost” in the form of CO_2 emissions during the production and application of these products. Irrigation of arid and semiarid lands can also increase a sink for SOC, but its contribution to a sink for C in soils must be discounted by CO_2 that is emitted when energy is used to pump irrigation water.

8.5 Summary

Turfgrasses exhibit significant C sequestration (0.34–1.4 Mg ha^{-1} $year^{-1}$) during the first 25–30 years after turf establishment. Several studies have reported that residential turfgrass soil can hold up to twofold higher SOC concentration than agricultural soils.

Published research suggested that dynamics of N is controlled by C transformation. Turfgrass areas have high levels of SOC and microbial biomass creating a C-based "sink" for inorganic N. Therefore lower than "expected" nitrate leaching and N_2O emissions have been measured in majority of the experiments carried out for turfgrass ecosystems. In turfgrass ecosystems, increased SOC can result from: (1) returning and recycling clippings, (2) an appropriate and efficient fertilization program, and (3) irrigating based on turfgrass needs to increase water use efficiency. Promoting SOC sequestration is one of the strategies for reducing atmospheric CO_2. However, to consider the net impact of urban grassland on the atmosphere's greenhouse effect, it is also pertinent to consider C costs of fuel and energy used in maintaining turfgrass, fertilizer and pesticide uses, and the fluxes of other GHGs mainly N_2O and CH_4 in addition to soil C sequestration. More research is needed to develop best management practices to achieve adequate production (presumably adequate turf quality) and minimize negative impacts (C cost, N_2O emission and nitrate leaching), thereby increasing turfgrass system's sustainability.

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Chapter 9
Carbon Sequestration Potential in Urban Soils

Sally Brown, Eric Miltner, and Craig Cogger

Abstract Urbanization is a world- wide phenomenon with expansion of urban areas often resulting in the loss of prime farmland and undeveloped land. In the U.S. developed land increased by 50% in the last 25 years. Urbanization can result in the loss of soil carbon (C) as a result of the expansion of impermeable surfaces, degradation of soils during construction, and lack of management. Impermeable surfaces can cover 80% of the soil surface in high- density urban areas, with this value decreasing to 30% when the greater populated area is considered. However, several factors suggest that there is a potential to increase C storage in urban areas. Research has shown that intensive management of urban soils can result in higher C reserves than similar soils in rural areas. Recently the importance of green-space in urban areas for storm water management, as a means to counter the heat island effect, and for restoration of limited ecological function has been recognized. Urban areas also generate large quantities of organic residuals that can be used as soil amendments. Over half of yard wastes generated are currently land applied, with approximately 50% of municipal biosolids and 98% of food waste landfilled or incinerated. Land application these amendments will accelerate C storage and can also replace synthetic fertilizers. Long- term studies in Tacoma, Washington found that 19–81% of amendment added C persisted in soils 3–18 years after amendment addition. Based on a conservative estimate using this data, application of residuals to pervious surfaces in Tacoma would result in an annual C sequestration rate of 0.22 Mg C ha^{-1} $year^{-1}$, similar to rates observed for no- till agriculture.

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Use of urban generated residuals offers a low impact and sustainable means to increase urban soil carbon reserves.

Keywords Biosolids • Compost • Organic residuals • Turfgrass • Land use

9.1 Introduction

Much of the research and discussion on the potential for soil carbon (C) sequestration has focused on C storage in agricultural or undeveloped lands. Urbanization, resulting in the destruction of ecosystems and prime agricultural land is occurring internationally at an accelerating rate (Boyle et al. 2010; Matteucci and Morello 2009; Pickett and Cadenasso 2009). For example, in the contiguous 48 States in the US 16.2 million ha of land were newly developed between 1982 and 2007 (USDA NRCS 2009). Of this, 2.3 million ha had been prime farmland. Urban growth is even more dramatic in the developing world with half of the world's population in urban areas. A majority of the predicted three billion population increase over the next 50 years is expected to reside in urban areas (Boyle et al. 2010). The large-scale growth of developed lands accompanied by the loss of traditional agricultural and forested lands indicate that it is appropriate to expand any discussion of soil C sequestration potential to include developed lands. This chapter will discuss the potential for soil C sequestration in urban areas. To provide background information for this topic, trends in urbanization are also discussed. This is followed by a discussion of characteristics of soils in urban areas, both as a function of end use and time since urbanization. Existing estimates of C sequestration in urban soils are presented. The chapter ends with a discussion on means to increase C stocks in urban soils in a sustainable manner.

9.2 Trends in Urbanization

The U.S. population doubled between 1950 and 2010. In 2004, about 80% of the 281 million people in the U.S. resided in urban areas (Alig et al. 2004). Growth of urban areas has generally increased more rapidly than changes in population (USDA NRCS 2009) with increased urbanization resulting in decreases in cropland and forestland. As of 1997, 5.2% of the total land area in the contiguous 48 states was classified as urban or developed land. The percent of land classified as urban as well as the population density and growth rate of urban areas varies by region. For areas with strict zoning, increases in population are not necessarily accompanied by increased land area being classified as urban. In the Portland, OR region, for example, population increased by 32% in the 1990s but land area classified as urban increased by only 22%. This trend was reversed in the Southeast. In Charlotte, NC, population increased by 33% while urban land area increased by 44%. The portion of land classified as urban is projected to increase

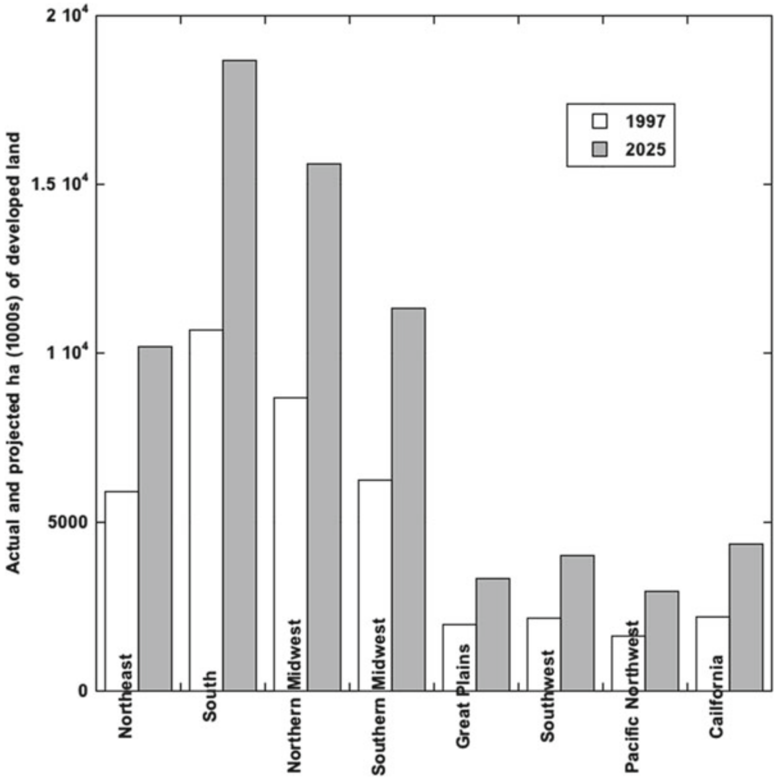

Fig. 9.1 Actual and projected hectares of developed land in different regions of the United States. Projected ha of developed land are middle range values (Data from Alig et al. 2004)

for all regions of the contiguous U.S. in the coming years. The quantity of land classified as urban (1000s of ha) as well as the predicted growth of urban lands by region is shown in Fig. 9.1.

9.2.1 Evapo-Transpiring Surfaces in Urban Areas

Urbanization has traditionally been associated with concrete and asphalt rather than trees and grass. Even in older urban areas however, the total portion of evapo-transpiring surfaces can be significant. For example, approximately 20% of the urban core of Manchester, UK consists of vegetated and water surfaces (James and Bound 2009). This figure increases to 72% when the area is increased to include Greater Manchester or 59% of what is considered the 'urbanized area' (Gill et al. 2007). Similar results have been reported for other regions in the United Kingdom

Fig. 9.2 Green roof picture. A green roof in Tacoma, WA. The soil material for this roof was made from urban organic residuals including municipal biosolids and woody debris (Photo credit: Dan Thompson)

(Stewart et al. 2008). Increasingly, urbanization in developed nations has included construction of low density housing with more open space than traditional urban cores. For example, low-density development or exurban development is occurring in the perimeter of Buenos Aires, Argentina (Matteucci and Morello 2009). While the sustainability of low density development is a concern, large scale areas that have been developed in this manner offer a high ratio of evapo-transpiring surfaces in relation to impermeable areas.

There is also an increasing understanding of the importance of incorporating sustainable infrastructure in built environments (Boyle et al. 2010). Factors driving this include climate change, increased urbanization, and resource availability. Water shortages and infrastructure for water treatment and solid waste disposal are primary considerations for the development of sustainable infrastructures (Boyle et al. 2010). Increased soil C reserves factor indirectly into these considerations. Urban planners are increasingly recognizing the importance of green space in urban areas as a means to counter the urban heat island effect, increase storm water infiltration, restore ecological function, and provide health benefits to urban residents (James and Bound 2009; Tzoulas et al. 2007). Higher C soils have been demonstrated to increase both water infiltration rates and water holding capacity in soils (Khaleel et al. 1981). The potential to convert impermeable surfaces to include limited ecosystem function through the construction of green roofs (Fig. 9.2) and living facades for buildings suggests additional opportunities for C storage in urban areas. For example, 20,000 ha in the UK consist of buildings with roofs that could be vegetated with little to no modification (James and Bound 2009).

In order to assess the potential for C sequestration on developed lands, several factors have to be considered. The C storage potential is likely to vary based on a range of factors including land use and distance from the urban core (Pickett and Cadenasso 2009). The quantity of open space in these areas will vary by age of development, population density, and other factors (James and Bound 2009; Tzoulas et al. 2007). The quantity of C in urban soils will also vary based on vegetative cover, maintenance history, and, for new developments, regulations and practices for topsoil restoration.

9.3 Soils in Urban Areas

9.3.1 Soil Formation Processes in Urban Areas

Soil formation and ecosystem function in urban areas differ from undisturbed sites. For example, water relations in urban soils are likely to differ from undisturbed areas. Infiltration rates for water in urban soils may be decreased as a result of compaction and sealing. Runoff from storm events is generally higher as a result of multiple impermeable surfaces that drain to storm sewers. Both of these factors lead to drier soils in urban areas in comparison to undisturbed landscapes (Hough 1995). In contrast, for certain urban landscapes, irrigation systems provide water in excess of what is available from natural sources (Brazel et al. 2000). Typical soil processes including alluvial deposition of sediments and fluctuating water tables resulting in alternating aerobic/anaerobic conditions do not generally occur in urban soils (Groffman and Crawford 2003). Nutrient budgets in urban systems may also differ from those in undisturbed systems. Many urban landscapes are intensively managed, resulting in significantly greater nutrient input than undisturbed systems. Conversely, abandoned or derelict sites are generally nutrient deficient. Urban landscapes also have unintentional nutrient inputs including nitrogen oxides (NO_x) from auto exhaust. Urban soils tend to have higher pH than comparable soils from non-urban environments as a result of decomposition of concrete. Urban soils also have higher temperatures than soils under similar management in more rural areas (Savva et al. 2009).

An extensive discussion of soils/soil formation processes and human impacts on these processes in Pickett and Cadenasso (2009) (Fig. 9.3) makes it clear that standard understanding of the factors of soil formation are not broadly applicable to urban soils. Drastic disturbances including fires, construction projects, storm sewers, impermeable surfaces, and intensive management have a much greater impact on soils than the traditional factors of soil formation. As a result, it is not necessarily appropriate to consider soil C sequestration in traditional terms. It is

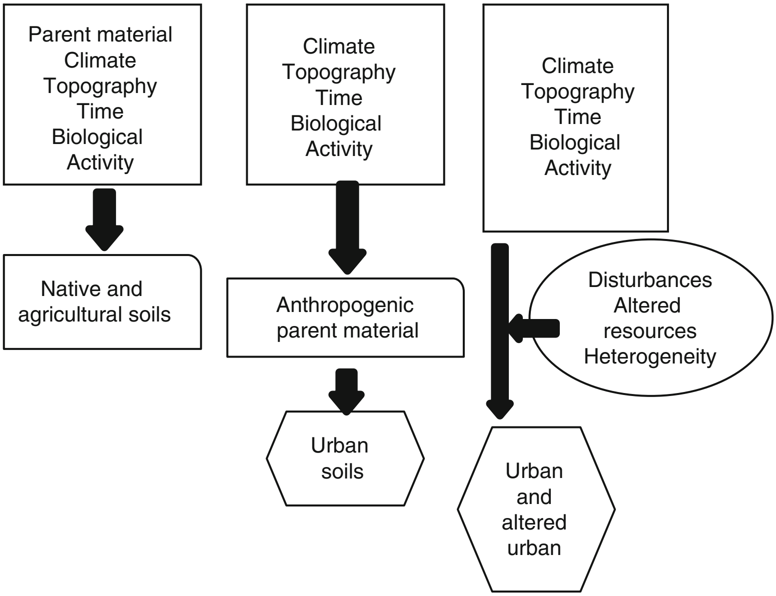

Fig. 9.3 A model to describe factors of soil formation in urban areas compared to soil development as a result of natural processes. The model shows the effects of both anthropogenic parent material as well as disturbance on soil formation (From Pickett and Cadenasso (2009))

Table 9.1 Mean, standard deviation, minimum and maximum values for select variables of urban soils collected from roadsides in Hong Kong (Jim 1998a, b)

		Mean	SD	Minimum	Maximum
Bulk density	$Mg\ m^{-3}$	1.66	0.25	1.15	2.63
pH		8.65	0.54	6.77	9.95
Organic Carbon	$g\ kg^{-1}$	8.1	8.4	0.8	40.1
Total N		0.4	0.8	0	4.8
Total Pb	$mg\ kg^{-1}$	113	75	12	420
Total Zn		122	96	38	504
Total Cd		1.2	0.6	0	2.75

more appropriate to consider how intensive inputs and management can increase soil C reserves for systems that have traditionally been marked by drastic disturbances.

9.3.2 Characteristics of Urban Soils

Variability and rapid change as a result of anthropogenic activities are the two primary characteristics of urban soils (De Kimpe and Morel 2000). Common

Table 9.2 Soil characteristics for urban soils from smaller urban areas. Means followed by the same letter are not significantly different

Parameter		Old residential	New residential	Old park	New Mulch	Mid Mulch
Bulk Density	Mg m^{-3}	1.41 bc	1.73 a	1.39 c	1.55 abc	1.59 ab
pH		6.93 a	7.08 a	6.73 a	6.91 a	6.64 a
P- Bray	mg kg^{-1}	132 ab	59 b	166 a	143 a	114 ab
Soil organic matter	kg m^{-2}	15.7 a	10.6 c	15.9 a	13.5 ab	11.8 bc

Data from Scharenbroch et al. (2005)

characteristics are often areas of high compaction, abrupt physical and chemical barriers to rooting depth and contamination by construction debris (De Kimpe and Morel 2000). Two surveys of urban soils in Hong Kong reported no evidence of pedological horizons (Jim 1998a, b). Instead, abrupt horizon changes were observed in many of the sites studied with random dumping of different types of debris. All sites were calcareous with elevated concentrations of Pb, Zn and Cd. Soils were generally highly compacted, and low in organic C and nitrogen (N). The mean ± standard deviation, minimum and maximum observed values for several of the measured variables are shown in Table 9.1.

The soils sampled for these studies represent an extreme example of urban soils. They were collected from a very densely populated urban core. For one sampling effort, the overlying concrete was removed prior to sampling. In the second study, samples were collected from $1 \times 1\ m^2$ sections that had been designated as sites for tree planting. Mediating factors such as fertilization and presence of plants that would be expected to improve soil physical properties were absent from these sites. It is likely that natural soil development had been stopped as a result of anthropogenic activities decades or centuries before samples were collected. In another study, anthropogenic disturbances led to topsoil burial under overlying horizons (Lorenz and Kandeler 2005). Soil sampled from a high density area, a park area and in proximity to a railroad in Stuttgart, Germany had significantly higher organic C concentrations at depths > 60 cm than at the soil surface.

Other studies to characterize urban soils have found different results as a consequence of time since disturbance, density of development, distance from the urban core and management practices. For example, Pouyat et al. (2007) sampled soils with different land use and cover in Baltimore, MD. In contrast with Jim (1998a, b), none of the soils sampled were in the inner urban core. They found significantly lower bulk density (P_b) in soils under forest cover in comparison to those under commercial or transportation uses. There was also a trend (not statistically significant) for greater soil organic matter (SOM) in forest and park soils in comparison to those under commercial and transportation uses. For this study, parent material was a significant factor in soil characterization with samples collected from soils derived from the Piedmont Plateau having different characteristics from those derived from Atlantic Coastal Plain parent material. This trend suggests that for less disturbed urban soils, or soils from more recently urbanized areas, traditional factors of soil formation have an impact on soil characteristics.

Scharenbroch et al. (2005) collected soil samples from vegetated sites representing an age chronosequence from small urban areas with low population densities (882 and 1,069 people km^{-2}). Older sites ranged in age from 52 to 102 years, newer sites from 1 to 5 years, and mid range sites from 15 to 20 years (Table 9.2). Land uses included residential, parks and mulched beds. The sampling showed that over time, P_b decreased while SOM and nutrients increased. Of particular note in this sampling were the values observed for samples collected from new residential areas. These samples are likely representative of what would be found in a wide range of soils in newer housing developments. As a consequence of increased urbanization with low population densities, suburban and exurban sprawl has resulted in large-scale construction of single- family homes. Standard construction practices often do not include stockpiling topsoil or use of composts to enrich soil following construction (http://www.sustainablesites.org/). The results from this sampling can be used as a general estimate of soil characteristics in newer residential, low- density developments.

Pouyat et al. (2002) studied soil C pools for different urban areas, land uses, and parent materials. Urban areas for this study were considerably larger with higher population densities than in the Scharenbroch et al., study (2005). The C density (total C to a specified depth per m^2), ranged from 1.6 to 28.5 kg m^{-2} for urban soils in New York City that had developed from non- native parent materials. These parent materials included refuse, dredged materials and clean fill. In contrast to the findings of Scharenbroch et al. (2005), C storage for low- density developments measured in this study was higher than for other land uses including forest, medium and high density and transportation. It is important to note here that the number of sites sampled in this category was small (n = 3) and no information was provided on the site age or site management practices.

9.3.3 Urban Rural Gradient

Several studies have assessed changes in soil characteristics as a result of urbanization using an environmental gradient approach from urban centers outwards. This type of assessment is important in the context of growing urban areas with low density development. In general, these studies have found increased C pools in urban soils in comparison to rural areas, with associated increases in biomass, respiration rates, trace gas fluxes and net primary productivity (NPP). These changes were likely a result of a combination of factors including more intensive management (increased water and nutrient input) and indirect impacts of proximity to urban environments including higher temperatures and increased N deposition. In one study of unmanaged creosote bush (*Larrea tridentate)* communities along an urban-rural gradient in Phoenix, AZ, soil C, and N increased in samples collected from the urban core area in comparison to the suburban and rural areas (Koerner and Klopatek 2010). In contrast to the rural and suburban sites sampled, these increases were seen both under plant canopies as well as in un-colonized areas. The increases in C and N were accompanied by decreases in respiration. Soil inorganic C (SIC) concentrations were also increased. Similar increases in urban soil C storage were observed by Pouyat et al. (1995, 2002). Undisturbed forest soils in an urban rural gradient in the New York metropolitan area were sampled. The C density was about 30% higher in the urban forest soils in comparison to the suburban and rural forests. A higher number of worm species, including non- natives, higher temperatures (2–3°C), increased heavy metal and salt concentrations also characterized the urban forest soils in comparison to the suburban and rural sites. Higher C storage in the urban sites was potentially associated with reduced degradability of litter material.

9.4 Urban Soil C Sequestration

Attempts to quantify the potential for soil C storage in the U.S. have not consistently included urban soils in their analysis (Houghton et al. 1999; Pouyat et al. 2006). Although rapidly increasing, the portion of land currently classified as high density in the U.S. is small (USDA-NRCS 2009). In addition, a paucity of data on urban soils as well as high variability in urban soil C content potentially reduces the accuracy of any estimates. The most thorough effort to carry out such an estimate was done using data from the published literature in combination with original sampling (Pouyat et al. 2006). The authors used data from six cities where information was available on soil C content and P_b for a range of end uses. These end uses included park, residential and clean fill. For their estimate, the authors assumed minimal C concentrations below a 1-m depth, low variability within each end use and that soil C concentrations were in equilibrium. They also included C content of soils under impervious cover. This estimate was extended to urban areas across the U.S. Types of land cover were estimated using the National Land Cover Database. The authors concluded that for cities where hardwood forests had been the dominant cover, urbanization resulted in a net decrease of soil C. For areas characterized by lower value forest and generally lower soil C pools, urbanization increased soil C reserves. Soil C reserves for the six cities, likely C reserves prior to urbanization for forest and agricultural soils, and the net change in C storage are shown in Fig. 9.4.

The authors estimated that soils in urban areas in more arid and warmer regions including the Southwest, South, and California will have a net increase in C storage as a result of urbanization. Higher levels of inputs including water and fertilizers would be expected to increase primary productivity and increase soil C reserves. They also estimate that C reserves may increase as a result of urbanization for areas with lower density where residential lawns occupy a significant area. Turf grass can

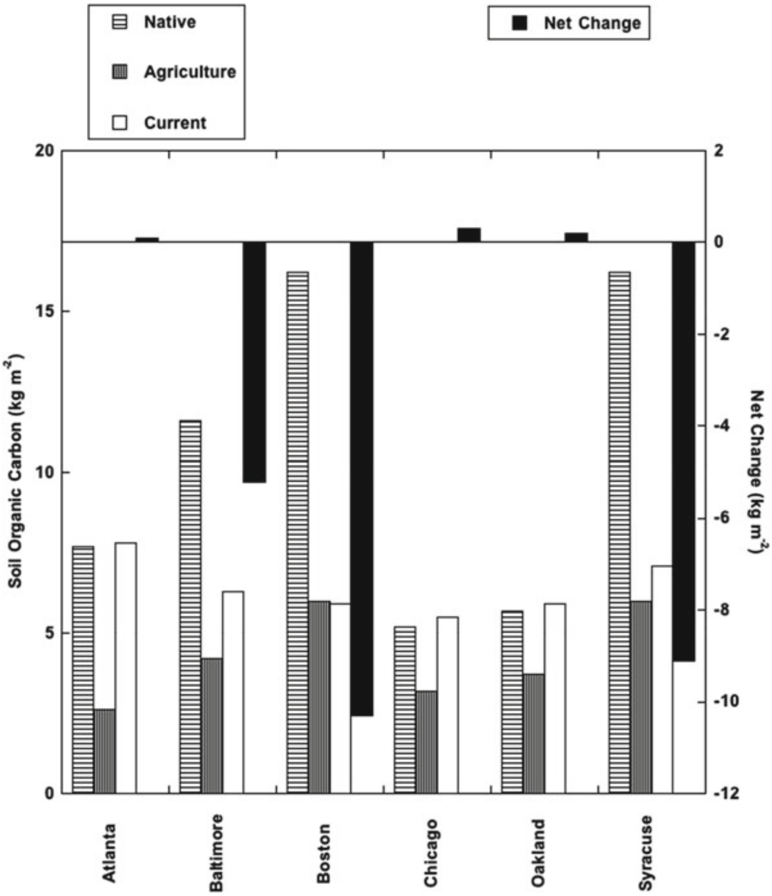

Fig. 9.4 Estimated soil organic C (kg m^{-2} to a 1 m depth) concentrations in 6 cities in the U.S. Native and agricultural soil C contents represent likely soil C concentrations for each region in the absence of urbanization. The net change in soil C as a result of urbanization was calculated assuming that land use in each area was equally divided between native and agricultural uses prior to urbanization (Data from Pouyat et al. 2006)

accumulate soil C at rates equivalent or higher to native grasslands and certain forests. The authors note the importance of accounting for C reserves under impervious surfaces in their estimates.

It is clear from the previous discussion that the fraction of total land area under urban development is increasing both within the US and worldwide. This urban expansion in many cases is at the expense of prime agricultural and forest lands. The nature of future urban development will vary based on population densities among other factors. It is also clear that soils in urban areas have very different characteris-

Box 9.1 Turf Grass

Turfgrass Contribution

As an important part of the urban landscape, turfgrass can make a significant contribution to C sequestration. As turf establishes and grows, C accumulates through the development of roots, thatch, and shoot tissue. Grass root systems can be either annual or perennial in growth habit, depending on species (Beard 1973). Aging roots die and slough off as new ones are added, contributing to C accumulation in the soil. Shoot tissue is constantly recycled into the canopy through regular mowing. Although grass clippings are predominantly water by weight, significant structural carbohydrates are returned to the plant canopy. Thatch has been defined as "a tightly intermingled layer of dead and living stems and roots that develops between the zone of green vegetation and the soil surface" (Beard 1973). Although it is generally not considered to be a component of SOM, thatch can impact surface SOM content as it is broken down and incorporated into the soil through foot or vehicle traffic, cultivation equipment, or earthworm activity. Because turf is managed as a perennial and the developed sod often remains intact for years or even decades, accumulated C is not subject to the transformational pressures that cultivated agricultural soils face. The potential for turfgrass to contribute to C sequestration has been well-documented.

Qian and Follett (2002) analyzed soil samples collected from 15 golf courses in Colorado and one in Wyoming, ranging in age from 1.5 to 45 years. Soil textures ranged from sandy loam to clay loam. They found that soil C increased at a rate of 0.9–1.0 Mg C ha^{-1} year^{-1} for approximately 31 years before the rate leveled off. They compared management practices at the various sites and found no relationship with fertilizer Bandaranayake et al. (2003) used the CENTURY model to predict changes in soil C in the upper 20-cm of soil beneath a golf course fairway turf. Their results indicated the C accumulation rates of 0.9–1.2 Mg C ha^{-1} year^{-1} for 30–40 years. The predicted results were validated with soil data from Qian and Follett (2002).

Porter et al. (1980) collected soil samples from approximately 100 turf grass sites in New York, varying in age from 1 to 125 years. The sites encompassed golf courses, cemeteries, and church lawns. They found that total N continued to accumulate for approximately 30 years, the rate of accumulation was the highest in the first 10 years. Although they did not measure C, if it is assumed that soil C and N were reasonably correlated, their accumulation curve was similar to that of Qian and Follett (2002) and Bandaranayake et al. (2003).

Milesi et al. (2005) estimated that there were approximately 164,000 ha of turfgrass in the U.S. using data from 2000 to 2001, or approximately 1.9% of the total land surface area. This makes turf the largest irrigated crop in the

(continued)

Box 9.1 (continued)

country, covering about three times the area of corn (*Zea mays L*). Assuming a uniform input of N of 147 kg ha^{-1} $year^{-1}$, they calculated sequestration of 5.9 Tg C $year^{-1}$ when grass clippings were removed following mowing, and 16.7 Tg C $year^{-1}$ when clippings were recycled into the turf. This is equivalent to a range of 0.36–1.0 Mg C ha^{-1} $year^{-1}$. For turf receiving no fertilizer or water, their estimate was a −0.2 Tg C $year^{-1}$, or −0.1 Mg C ha^{-1} $year^{-1}$, a net loss of C. Although Qian and Follett (2002) found no relationship between N fertilization and C sequestration, N fertilizer is generally applied to golf courses, and presumably was applied to the sites in their study, although it was not reported. The results of Milesi et al. (2005) illustrate the impact of moderate N application on C sequestration, and the importance of returning clippings to the turf canopy during mowing.

Townsend-Small and Czimczik (2010a) measured organic C in soils beneath turf in parks ranging in age from 3 to 24 years in Irvine, CA. They found an accumulation rate of 1.4 Mg C ha^{-1} $year^{-1}$. They also computed the hidden C costs of maintaining turf, estimating CO_2 emissions from fuel consumption, irrigation, and fertilization. This estimation resulted in net CO_2 consumption of approximately 1.1 Mg CO_2 ha^{-1} $year^{-1}$. An estimation, taking into account emissions related to system inputs, resulted in net sequestration of 0.3 Mg C ha^{-1} $year^{-1}$ (Townsend-Small and Czimczik 2010b).

Several studies have addressed the issue of C sequestration by turfgrass. Their estimates of gross annual rates are similar, ranging from 0.3 to 1 Mg C ha^{-1} $year^{-1}$ for up to 30 years after establishment, before leveling out. Turfgrass is a widespread component of the urban/suburban landscape that offers significant potential to contribute to C sequestration.

tics from those that have developed without anthropogenic influence. Urban soils are potentially best categorized by their heterogeneity, with widely variable characteristics based on parent material, management and proximity to and age of the urban core. For undisturbed sites, studies that have looked at soils along an urban rural gradient have generally reported higher metal concentrations, electrical conductivity and C in urban in comparison to rural soils. For disturbed sites, depending on the end use and parent material, C storage can vary within the same municipality by a factor of 10. Studies have also shown that soils with a vegetative cover, within a particular region and end use, may have increasing C concentrations over time (Scharenbroch et al. 2005). For certain regions, it is recognized that intensive management of urban soils can result in increased NPP and soil C sequestration (Koerner and Klopatek 2010; Pouyat et al. 2006). However, the C costs of intensive management need to be factored into any C accounting.

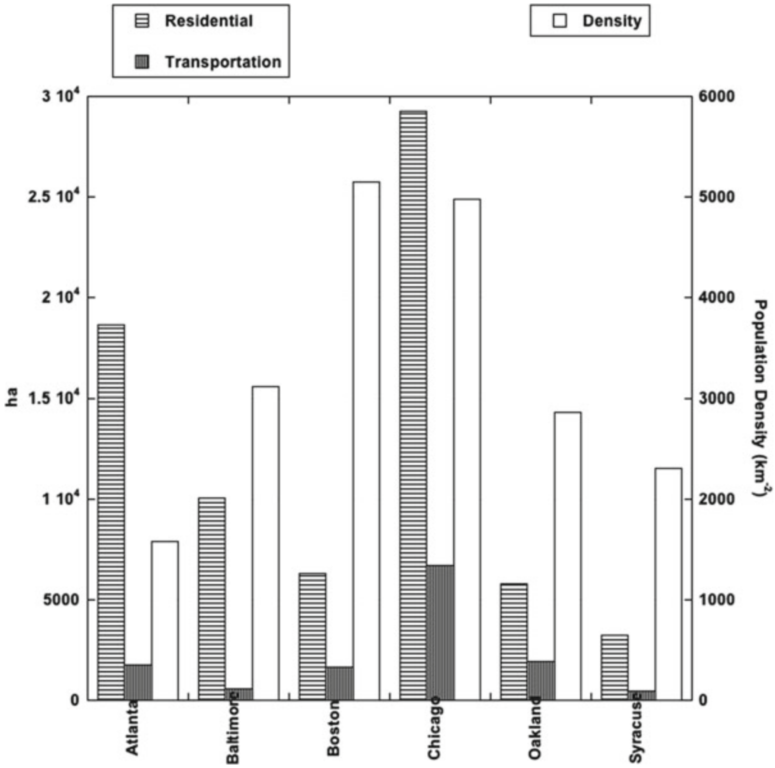

Fig. 9.5 Population density (individuals ha^{-1}), and amount of land in residential and transportation related uses for select cities in the U.S (Data from USDA NRCS 2009)

There are additional factors unique to urban areas in the U.S. that suggest that the potential capacity for soil C may be greater than anticipated. Demographic trends in the U.S. towards increased low- density development indicate that a high portion of newly urbanized areas will consist of open space rather than impermeable surfaces. Population density and land area for residential and transportation end uses for a number of municipalities in the US are shown in Fig. 9.5.

Atlanta, the city with the lowest population density in this figure, is also the city with the highest predicted population growth (Fig. 9.1). Typical low- density residential developments consist of 3.8 homes per ha with average occupancy of 2.6 individuals per home for a density of 1,000 people km^{-2} (Pozzi and Small 2001). Development at this density leaves the area with vegetative cover over 20–70% of the land surface with 30–80% of the surface considered as 'constructed' with either residential structures or road covering the soil surface (Vogelmann et al. 2001).

Conventional residential construction practices are often highly destructive of both native vegetation and topsoils. In many ways, construction of new residential developments has similar effects on soil function as surface mining operations (Akala and Lal 2001). Building practices in certain regions currently require use of organic soil amendments as a means to rapidly restore soil function (Stenn 2010). This suggests that recent home construction in urban areas has resulted in the loss of soil C with an associated potential soil C sink.

In older urban areas, many abandoned former industrial sites are now being converted to green space. The US EPA Brownfields program currently has 1,494 properties that cover 5,655 ha ready for redevelopment (Ann Carroll (2010), US EPA, Office of Brownfields and Land Revitalization, carroll.ann@epa.gov, personal communication). There are a total of 24,450 ha in 150 cities that are considered to be Brownfields. Of the sites that are being redeveloped, 62% include land for use as parks or open space (United States Conference of Mayors 2010). This is likely to involve conversion of impervious cover to highly managed park, native or residential land. The importance of green-space in urban areas is highly beneficial for a range of factors including human health and well being and ecosystem health (Tzoulas et al. 2007).

Finally urban areas produce large quantities of organic residuals that have the potential to be used, either directly or after treatment as soil amendments. Local use of residuals offers a more sustainable option than landfilling or combustion of these materials. Intensive use of organic soil amendments can greatly accelerate the rates of soil C sequestration while simultaneously increasing NPP. These amendments can also supply sufficient nutrients for plant growth, supplanting the need for synthetic fertilizers with a resulting decrease in emissions associated with the manufacture of these materials (Brown et al. 2010). Using these amendments in place of traditional fertilizers has the potential to result in credits due to both increases in soil C reserves and offsets from avoided fertilizer use. Data from the City of Tacoma, WA can be used to illustrate the potential for urban C sequestration using municipal residuals.

9.4.1 Organic Amendments – Tacoma, WA Case Study

9.4.1.1 Supply of C-Rich Amendments in Tacoma, Washington

The City of Tacoma in western Washington State has a population of 198,000, total land area of 130 km^2 and population density of nearly 1,500 people km^{-2} (US Census Bureau 2011). Developed residential lots cover 31% of the City's land area (4,000 ha) and parks cover 8% (1,200 ha) (City of Tacoma 2006). The majority of the dwelling units in Tacoma are single-family detached houses (61%) (US Census Bureau 2009). About 45% of the city is covered with impervious surfaces (USGS 2010). The population of Tacoma grew at a rate of 2.5% between 2000 and 2009 although more slowly

Table 9.3 Annual generation of biosolids, yard waste, and food waste in Tacoma, WA, and potential C sequestration 7–15 years after application to urban landscapes

		Annual generation raw materials	Annual production after processing	Amendment C after processing[c]	Potential C sequestration[d]
Material	Processing	Mg	Mg	Mg	Mg
Biosolids	Blend	3,600	6000[a]	1,920	460
Yard waste	Compost	12,500	7500[b]	2,400	575
Food waste	Compost	10,000	6000[b]	1,920	460
Total		26,100	19,500	6,240	1,495

[a]Includes 2,400 Mg wood product mixed with biosolids

[b]Based on 40% loss of dry mass during composting (Sommer and Dahl 1999; Barrington et al. 2002; Hao et al. 2004)

[c]Based on mean 32% C concentration in amendments

[d]Estimated as 24% of C applied, based on Table 9.4

than the surrounding suburbs, which grew 13.5% over the same time period (US Census Bureau 2011).

The Tacoma wastewater treatment plant generates 3,600 dry Mg year^{-1} of biosolids and residents recycle 12,500 dry Mg year^{-1} of yard debris to commercial composters (City of Tacoma, 2010, Dan Thompson, Environmental Services/ Waster Division Manager, Personal communication). Annual food waste production is estimated at 10,000 dry Mg year^{-1} (Fuchs and Frear 2006). This is equivalent to 18 kg biosolids, 63 kg yard debris and 50 kg food waste per person per year. The biosolids meet Class A pathogen reduction standards through a thermophilic treatment process, and are mixed with wood products and sand to produce soil amendments for landscaping and gardens (marketed as Tagro products). The yard debris is composted to make a soil amendment. Most of the food waste is still disposed of in landfills, but future options include direct composting or anaerobic digestion followed by composting. During the composting process a portion of the raw organic materials are decomposed, but a substantial amount of C remains after processing. Actual production of yard waste compost and potential production of food waste compost total more than 13,000 Mg year^{-1} finished product from materials originating within the City, and contain more than 4,000 Mg of C (Table 9.3). Actual production of biosolids blends is about 6,000 Mg/year of organic material or 1,800 Mg/year C, including the biosolids and wood products added as blend ingredients (Table 9.3).

Based on changes in soil C concentration determined from long-term (7–15 years) soil amendment studies (See box Table 9.5), nearly 1,500 Mg of the 6,240 Mg C in the composts and biosolids that are produced annually and land applied can be sequestered into the SOM pool.

On a land area basis, the production of processed organic amendments is equivalent to 1.5 Mg ha^{-1} year^{-1} dry weight over the total land surface or 2.7 Mg ha^{-1} year^{-1} over the pervious surface within Tacoma city limits. Based on the total C content of the amendments and the fraction of total C likely to remain in the soil, this is equivalent to an annual C sequestration rate of 0.12 Mg C ha^{-1} year^{-1} across the municipal land area and 0.22 over the portion of the City covered by pervious surfaces. These rates of C sequestration are similar to those reported for conversion from conventional to no-till agriculture (0.308 ± 0.280 Mg C ha^{-1} year^{-1}, Spargo et al. 2008). Considering the amount of amendment produced, and the C sequestration potential (shown in replicated field trials using these amendments), it is clear that locally produced organic materials can have a significant effect on soils and C sequestration in urban areas.

9.4.2 Demand for C-Rich Amendments in Tacoma, Washington

Tagro products made from Tacoma biosolids are primarily used for amending residential gardens and landscapes and commercial landscapes in Tacoma and the surrounding suburbs. Demand exceeds supply, resulting in 6,000 Mg of amendment applied and an estimated 460 Mg of C sequestered each year (Table 9.3).

Amendment of disturbed soils following new construction is now required in the Puget Sound area as part of best management practices (BMPs) for stormwater management (Stenn 2010), creating additional demand for organic amendments. This market is more suitable for compost than Tagro products, because Tagro is too N-rich to be applied at the prescribed rates. Default application rates to comply with the BMPs are 7.5 cm of compost for landscape beds (approximately 225 Mg ha^{-1}) and 4.5 cm of compost for turf areas (approximately 130 Mg ha^{-1}). New construction with 50% of the green area planted to turf and 50% planted to landscape beds would require about 175 Mg compost per hectare of green area at these rates. One year's production of Tacoma compost would be enough to amend about 80 ha of disturbed soil. Construction of new single-family homes has averaged 350 units per year in recent years (City of Tacoma 2010), which would consume less than 12% of the amendments produced, assuming a green area per lot of 0.025 ha. Total single-family home construction in Tacoma and surrounding urban growth area has averaged 3,000 units per year (Carlson and Dierwechter 2007), which would require about 13,000 Mg of amendments, or nearly all of the production of Tacoma compost. Additional current uses of compost include amendment and improvement of existing home landscaping and public parks, urban agriculture, commercial and roadside landscaping, renovation of disturbed soils, and construction of rain gardens for bioretention of stormwater (Fig. 9.6). It is clear that all of the soil amendments currently and potentially produced from Tacoma's urban organic wastes could be used to improve local urban soils, sequestering an estimated 1,500 Mg of C year^{-1}.

Fig. 9.6 Urban agriculture pictures. The growth of urban agriculture is one example of increasing green space in urban areas. Here community gardens are being developed on abandoned lots in Seattle, WA (Photo credit: Sean Conroe. Soil is being amended with a biosolids compost in preparation for planting. Photo credit: Becky Warner)

Box 9.2 C Storage in Amended Soils

Research conducted in western Washington State provides estimates of C sequestration potential from soil amendments during transition from agricultural to urban land. Soil C was measured in five soil amendment experiments, three that shifted from row crops to grass, one that shifted from row crops to a woody landscape typical of residential and commercial landscape beds, and one on a disturbed urban soil along a highway roadside that was converted to urban landscaping (Table 9.4, Fig. 9.7). Soil C levels in the unamended treatments ranged from moderately depleted to moderately high. The highway soil was also highly compacted with a P_b of 2 Mg m^{-3}. The soils used for these studies were generally higher in organic C compared with both highly degraded urban soils as well as managed soils from urban forests and lawns (Jim 1998a, b; Pouyat et al. 2002; Scharenbroch et al. 2005). No tillage was done in any of the experiments after establishing the plantings.

Treatments included a range of soil amendments and rates (Table 9.5). A single amendment application was made at four of the sites, while one site received ten annual applications. Soil C was measured in 2008, 7–15 years after the agricultural transition sites were established, and 1 year after the urban transition site was established.

Table 9.4 Description of studies at urban transition experimental sites including soil series, crop, age and unamended soil carbon concentration

Site	Soil series	Taxonomy subgroup	Unamended soil C g kg^{-1} soil	Planting	Year established	Reference
Fescue compost	Puyallup	Vitrandic Haploxerolls	22	Tall fescue	1993	Sullivan et al. (2003)
Fescue biosolids	Puyallup	Vitrandic Haploxerolls	20	Tall fescue	1993	Cogger et al. (2001)
Home Turf	Briscot	Aeric Fluvaquents	35	Perennial ryegrass	2000	Unpublished
Landscape	Puyallup	Vitrandic Haploxerolls	10	Mixed woody landscape	2001	Cogger et al. (2008)
Highway	Urban land (compact glacial outwash)	–	13	Mixed woody landscape	2007	Unpublished

(continued)

Box 9.2 (continued)

Fig. 9.7 Installing an experiment to test plant response to compost amendments along a highway right of way outside of Tacoma, WA (Data from this study is shown in Table 9.5. Photo credit: Andy Bary)

The amount of C sequestered was similar across the four long-term sites, averaging 24% of C added in the amendments. Time series data collected from the tall fescue (Festuca arundinacea) compost site shows an initial rapid decline in soil C in the first 5 years after amendment, with a slight increase in C for the following 10 years. The high proportion of C retained in the disturbed urban soil at the Highway site reflects in part the short time between incorporation and sampling. But, comparison of C retention after 1 year at the Highway site (Table 9.5) with the fescue compost site suggests that the potential for C sequestration is as great or greater in the urban soil as in the former agricultural soils. These results suggest significant long-term C sequestration potential from soils amended with biosolids or compost, and managed as turf or landscape plantings.

(continued)

Box 9.2 (continued)

Table 9.5 Carbon sequestration at urban transition sites 2–15 years after experiments were established

Site	Amendment	Total amendment rate Mg ha^{-1}	Total carbon applied Mg ha^{-1}	Application Type	Years applied	Soil C increase Mg C ha^{-1}	Mean C sequestered as% of added C
Fescue compost	Food-waste compost	157	47	Incorporated	1993	9	19
Fescue biosolids	Class A biosolids	67–134	22–44	Surface	1993–2002	6–10	27
Turf	Yard-debris compost	74–224	19–57	Incorporated	2000	2–20	25
Landscape	Yard-debris compost	224	49	Incorporated	2001	13	26
Highway	Yard debris compost	150	65	Incorporated	2007	31	48
	Biosolids-wood mulch	147	57	Incorporated	2007	46	81

9.4.3 *Implications for Urban C Sequestration in the United States*

Approximately 58 Tg of biosolids, food waste, and yard waste combined are generated in the United States each year, with about a third of the total recycled via land application (Table 9.6). The great majority of these raw materials are generated in cities and their surrounding suburbs. The national per capita generation rate is slightly greater than the Tacoma data, but the current recycling rate of biosolids and yard debris is much lower than Tacoma's.

Capturing and composting all of the biosolids, food waste, and yard waste would supply enough organic material to spread across the 40 million ha of urban land in the United States at a rate of nearly 0.9 Mg dry matter ha^{-1} $year^{-1}$ assuming 40% mass loss during composting. If this organic material were to be applied to the land that is converted annually from rural to urban use (approximately 640,000 ha/year), the compost application rate would be about 54 Mg ha^{-1} $year^{-1}$. Assuming 50% of the land under urban conversion was used as green space, more than 100 Mg ha^{-1} organic material, equivalent to 3–4 cm depth, would be available for land application. This would be enough to substantially improve soil quality across all newly developed land. Potential C sequestration associated with this practice across urban areas of the United States would be nearly 3 Tg $year^{-1}$ (Table 9.7).

Table 9.6 Urban-based organic residuals generation rates in the United States

	Generation per person	United States total		Total recycled
	kg $year^{-1}$	Tg $year^{-1}$	% Recycled	Tg $year^{-1}$
Biosolids[a]	22	6.7	50	3.4
Yard debris[b]	88	26.8	55	14.8
Food waste[b]	79	24.1	2	0.5
Total	189	57.6		18.6

[a]Data from NEBRA 2006
[b]Data from US EPA 2006

Table 9.7 Carbon sequestration potential of urban-derived organic matter applied to urban soils in the United States

US organic waste generation	Net compost generation[a]	Mean C mass[b]	Potential C sequestration[c]
Tg $year^{-1}$			
57.6	34.6	11.1	2.7

[a]Based on 60% retention of mass during composting
[b]Based on 32% C in amendments (mean of amendment C concentration in Tacoma study)
[c]Based on 24% sequestration of applied C in soil organic matter (mean of Tacoma data)

9.5 Conclusions

Previous research has shown wide variability in the C content of urban soils. Soils in highly disturbed areas or recently built areas have significantly lower organic C contents than intensively managed landscapes. Managed urban landscapes in turn, often have higher organic C contents than corresponding areas outside of the urban boundary. This suggests a high C storage potential for a significant portion of soils in urban areas. Intensive management is one option to rapidly increase C reserves, at reported rates greater than 1 Mg C ha^{-1} $year^{-1}$. This can have a high impact in areas of lower population density or in urban cores with existing open space. However, intensive management using conventional fertilization and irrigation practices is associated with significant C emissions. Use of urban generated organic residuals provides a means to increase C reserves with minimal associated C emissions. Increasing soil C reserves in urban areas can have a small, but significant, positive impact on terrestrial C storage.

Another factor to consider is the indirect role of increased soil C reserves in the sustainability of built environments. Urbanization is occurring at a high rate across the globe. Most of the expected population growth over the next 50 years will reside in urban areas. This will result in challenges for sustainable infrastructure, water availability and residuals management. Many components of a sustainable urban infrastructure are dependent on functional soils. Increasing soil C reserves through construction of green roofs, bio-swales, road-side soil management will alleviate the urban heat island effect and result in reduced storm water flows and improve water storage. Increasing urban soil C reserves can also serve to create a local market for urban generated organic residuals, resulting in a more sustainable waste management infrastructure.

Increasing soil C reserves in urban areas has a dual purpose: as a means to increase soil C storage as a climate mitigation technique, and as a means to create more sustainable built environments.

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Chapter 10
Carbon Sequestration in Turfed Landscapes: A Review

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Abstract The objective of this chapter is to summarize the findings of the existing literature that studies carbon (C) flux in turfgrass systems. First, studies that evaluate C in the unique parts of the turfgrass system (thatch, specialized stems, clippings, verdure) are relatively scant, with the majority of those few studies focusing on C removal via clipping harvest. In the published literature, average estimates of C in turfgrass verdure (stubble), roots, and underlying soil organic C and inorganic C are 100, 139, 4,300 and 9,000 $g\,m^{-2}$, respectively. The degree of C storage in soil varies with the age of the turfgrass sward and its management, and only one published paper has examined differences due to grass species. The very few studies that attempt to quantify C flux in turfed systems indicate that mowing (including the harvest and removal of C in clippings) most affected C release from the turfgrass system. The impact of turfgrass management practices, especially mowing, N fertilization, and irrigation, on C sequestration is the area which needs greatest study. Such work is especially needed in long-term studies with a greater variety of turfgrass species.

Keywords Turfgrass • Lawn • Carbon accumulation • Thatch

10.1 Introduction

The American lawn is as old as the first European settlement. An English and French innovation, lawns were brought with the British to America (Bormann et al. 2001). Virtually all of our cool-season turfgrasses have European origins, and were brought

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to America as a food source for grazing animals. Our warm-season turfgrasses have a more diverse introductory path into the United States, and many (such as centipedegrass) have a traceable path to a sole plant collector (Cunningham 1984). The American urban landscape became a more common installation in the mid-1800s, increasing in popularity with the advent of the Industrial Revolution, the middle class, and the American suburb (Bormann et al. 2001).

Mechanization and changing careers (from agricultural to service-based) have long led to the rise of a middle class with increasing leisure time. For many, this leisure time is spent in sports, which often require large areas of turfed playing fields, or in the direct maintenance of a personal turfed space, such as a home lawn. With this comes an increase in population and numbers of households. For example, in the Indian County (USA) area it was estimated that there was a 7% increase in population from 1970 to 2000, with a 10% average annual increase in the number of households (Liu et al. 2003). Although population and the number of households increased, the average number of people in the household actually decreased (from 2.90 to 2.25), with a result of 11,103 extra households present in 2000, due to the reduction in average household size. It is likely that some portion of those extra households will have a grassed landscape. In this same year another research paper demonstrated that residential lot size (in Ohio) increased over the past century, and that the percentage of those lots that was potential lawn had increased to approximately 25% of the total urban land area (Robbins and Birkenholtz 2003). A third paper published in the same year found (using remote sensing) that in 1992/1993 urban areas were 4.5% of the total land area in the southeastern United States. By 2000 the percentage of land that was in urban development was estimated at 6.4% (Milesi et al. 2003). When two watersheds in Maryland were evaluated, the estimated percent lawn area was 15% and 25.5% for the two watersheds (Zhou et al. 2008). With a total estimate of 10–16 million hectares of lawn in the United States (Zhou et al. 2008; Robbins and Birkenholtz 2003) the contribution of lawns to C cycling could be substantial.

Although carbon dynamics in landscapes containing trees and or agricultural crops has received significant study in recent years, areas managed with perennial turfgrasses have had less evaluation. Soil and plant carbon (C) dynamics in urban settings are acknowledged to be different from those of forest or agricultural landscapes (Kaye et al. 2006), and turfgrass is often a primary part of such urban landscapes. When the impact of turfgrass on soil C is evaluated, it is often part of a larger study, and the turf is included within a home landscape, along with trees and shrubs. In other studies the impact of the turf itself is not quantified (via clipping analyses, etc.), and only the underlying soils are studied. In still other work archival or remotely sensed data are used to estimate carbon accumulation as affected by landscape, with an ‘urban’ landscape designation usually including a turfed area. Additionally, some carbon work has been completed in forage or pasture grasses, using species that may be closely related to those managed as turfgrasses (Haile et al.

2008; Sigua and Coleman 2010). The objective of this chapter is to provide a review of the published literature which has examined C in turfgrass systems, and to discuss further needed research in this area.

10.2 Turfgrass Species

For those who do not manage turfgrass on a regular basis, an introduction to the peculiarities of turfgrass management may be helpful. First, the species of turfgrass grown varies widely with geographic location, primarily a factor of cold tolerance. In the northeast, Midwest, and western United States primary turfgrass species are the 'cool-season' grasses, while in the southeast, south and southwestern United States 'warm-season' turfgrasses are commonly grown (Table 10.1). The terms 'cool' and 'warm' season refer to the temperature climes in which the grasses thrive. Cool season grasses are typically found in temperate and subarctic climates, and are long-day plants. Photosynthetically, they are C_3 plants (Turgeon 2008). Warm-season turfgrasses occur mostly in subtropical, tropical and warm climates, and their photosynthesis is through the C_4 pathway. They are often short- or intermediate-day plants that require warm nights (Turgeon 2008). In many warm (not tropical) regions cold nights and frost will send some warm season turfgrasses into dormancy, and straw-colored dormant turfgrasses are a common sight in many winter southeastern and southwestern lawns in the United States.

Turfgrasses have a variety of plant structures that may also affect C dynamics in the plant. For example, they may have specialized stems growing above (stolons) or below (rhizomes) ground (Fig. 10.1). These prostrate, spreading stems produce roots, tillers, and leaves, and are a storage organ for carbohydrates. Some warm-season turfgrasses such as zoysigrass (*Zoysia* spp.) or bermudagrass (*Cynodon* spp.) have extensive stolon and rhizome production. Other grasses, such as most of the ryegrasses (*Lolium* spp.), only spread via side shoots ('tillers') and are considered 'bunch-type' turfgrasses.

Specialized use of some cool season grasses such as perennial ryegrass for overseeding (*Lolium perenne* L.) and bentgrass (*Agrostis* spp.) for putting greens pushes their use farther south into regions where they are only marginally adapted. Likewise, experimental use of bermudagrass and zoysiagrass is found farther into the colder regions of the Midwest and northeast. Regardless, the term 'turfgrass' encompasses a great many grass species, many which differ in growth habit (prostrate versus spreading), growth period (warm season grasses will go dormant after a killing frost, for example) and physiological functioning (C_3 versus C_4). In the same way that there are no universal N fertilization recommendations for turfgrass, it is possible that findings about carbon (C) in turfgrass systems will be varied from species to species, region to region, and even within turfgrass use (e.g., athletic fields, lawns, putting greens).

Table 10.1 Species of turfgrass commonly grown throughout the United States, and their common designated uses in turfed ecosystems

Species – common name	Latin name	Primary use(s)	Other notes of interest
Cool season turfgrasses			
Creeping red fescue	*Festuca rubra* L. ssp. *rubra* Gaud.	Lawns, golf course fairways (Europe), roughs	
Chewings fescue	*Festuca rubra* L. ssp. *comutata* [Thuill.] Nyman	Lawns, golf course fairways (Europe), roughs	
Sheep fescue	*Festuca ovina* L. spp. *hirtula* [Hackel ex Travis] Wilkinson	Erosion control, utility turf	
Hard fescue	*Festuca trachyphylla* [Hackel] Krajina	Erosion control, utility turf	
Tall fescue	*Festuca arundinacea* Schreb.	Lawns, utility turf, low management athletic fields, municipal	Widely adapted
Meadow fescue	*Festuca elatior* L.	Utility turf	
Kentucky bluegrass	*Poa pratensis* L.	Lawns, fairways, high quality athletic turf, municipal	Widely adapted, not grown in southeast US
Texas bluegrass	*Poa arachnifera* Torr.	Native turf	
Canada bluegrass	*Poa compressa* L.	Erosion control	
Hybrid bluegrass	*Poa pratensis* L. x *Poa arachnifera* Torr.	Lawns, utility turf	A relatively new turfgrass in commercial production
Rough bluegrass	*Poa trivialis* L.	Overseeding warm season turf, especially putting greens	Often viewed as a weed in cool season areas
Annual bluegrass	*Poa annua* L. var. *annua* and *Poa annua* var. *reptans* (Hauskn.) Timm	Fairways, putting greens, a part of many cool season turfgrass populations	Largely viewed as weed in warm-season turfs
Supina bluegrass	*Poa supina* Schard.	Athletic fields (Europe)	Being evaluated at the experimental stage in U.S.
Perennial ryegrass	*Lolium perenne* L.	Fairways, athletic fields, overseeding warm season turf	
Annual ryegrass	*Lolium multiflorum* Lam.	Erosion control, overseeding warm season turf	
Intermediate ryegrass	*Lolium xhybridum* Hausskn.	Overseeding warm season turf	
Smooth bromegrass	*Bromus inermis* Leyss.	Erosion control, utility turf	

(continued)

Table 10.1 (continued)

Species – common name	Latin name	Primary use(s)	Other notes of interest
Creeping bentgrass	*Agrostis stolonifera* L.	Putting greens, fairways, tees	In the upper regions of the southeast US, only use is as a putting green turf
Colonial bentgrass	*Agrostis capillaris* L.	Putting greens	NW and NE U.S. only, Europe
Velvet bentgrass	*Agrostis canina* L.	Putting greens	NW, NE and upper midwest only, Europe
Redtop	*Agrostis gigantea* Roth.	Erosion control	
Crested wheatgrass	*Agropyron cristatum* [L.] Gaertn.	Native grass, roughs, lawns	U.S. Great Plains, prairies
Warm season turfgrasses			
Bermudagrass	*Cynodon dactylon* [L.] Pers.	Lawns, rough, utility turf.	Widely adapted
Hybrid bermudagrass	*Cynodon dactylon* [L.] Pers. X *Cynodon transvaalensis* Burtt-Davy	Fairways, tees, putting greens, lawns, athletic fields, municipal turf.	Widely adapted
African bermudagrass	*Cynodon transvaalensis* Burtt-Davy	Putting greens.	Not widely used in the U.S.
Zoysiagrass	*Zoysia japonica* Steud.	Lawns, fairways, tees, municipal turf.	
Zoysiagrass	*Zoysia matrella* [L.] Merr.	Lawns, fairways, tees, putting greens, municipal turf.	
Zoysiagrass	*Zoysia pacifica* (Goudsw.) Hotta and Kuroki	Lawns, municipal turf.	
Hybrid zoysiagrass	*Z. japonica* Steud. x *Z. pacifica* (Goudsw.) Hotta and Kuroki	Lawns, municipal turf.	
Carpetgrass	*Axonopus affinis* Chase	Lawns, utility turf.	Often viewed as a weed in the South.
Bahiagrass	*Paspalum notatum* Flugge	Utility turf, pastures.	
Seashore paspalum	*Paspalum vaginatum* Swartz	Lawns, fairways, putting greens, municipal turf.	Highly salt tolerant turf used in tropical climes.
Kikuyugrass	*Pennisetum clandestinum* Hochst. ex Chiov.	Fairways, lawns.	
Saint Augustinegrass	*Stenotaphrum secundatum* [Walt.] Kuntze	Lawns, municipal turf.	Southern and southwestern U.S.
Centipedegrass	*Eremochloa ophiuroides* [Munro] Hack	Lawns, municipal turf.	Low-input turfgrass
Buffalograss	*Buchloe dactyloides* (Nutt.) Engelm.	Lawns, low input turfs, municipal turf.	Adapted to semiarid regions.

Latin names are from the following references: Anderson et al. (2002), Barton et al. (2009), Gulse et al. (2007), McElroy et al. (2002), Patton and Reicher (2007), Turgeon (2008)

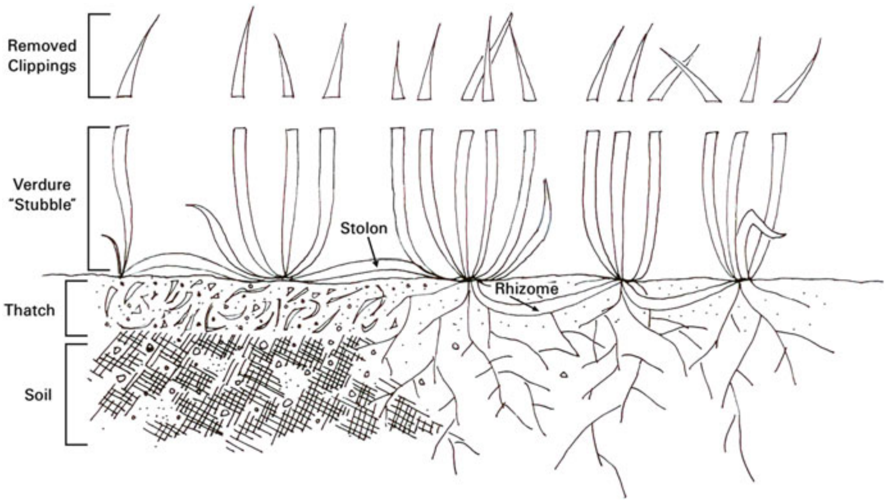

Fig. 10.1 A representative illustration of a turfgrass sward, showing clippings, verdure, specialized stems (rhizomes and stolons), thatch, and underlying soil

10.3 Management of Turfgrasses

Turfgrasses are managed with specialized techniques, and virtually all of these management strategies differ from those used in field or forest systems. In some cases even the propagation of turfgrasses is different, as many common warm-season turfgrasses are sterile hybrids (Table 10.1) and must be propagated via sprigs (planting of the specialized stem structures called rhizomes or stolons) or sod. Turfgrasses are a rapidly growing crop, and many grasses can double their leaf height in 2–3 days, making frequent mowing a part of turfgrass management. This frequency of mowing may be once a week on lawns or athletic fields, to every day on a highly managed putting green. In many cases these clippings are removed from the turf, a frequency of nutrient and dry matter removal not seen in any other crop. The removal or return of clippings could affect C flux, and is a unique facet of turfgrass management that deserves study. For example, returning grass clippings to a cool-season lawn increased dry matter yield and N use efficiency, and reduced N fertilization (Kopp and Guillard 2002). Similar results were observed in warm-season turf, where hybrid bermudagrass clippings decomposed rapidly over a 28 day period, with mineralization of 20% of clipping C over the incubation period (Shi et al. 2006a). Clipping removal/replacement will also affect C dynamics in the turfgrass left after clipping. Often called 'stubble' in the papers reviewed later in this Chapter, the more correct term often used in the turfgrass literature is 'verdure', which is used to refer to all above-ground plant tissue left after mowing and clipping removal has occurred (Fig. 10.1).

Fig. 10.2 Core aerification of turfgrass, a practice commonly done to relief soil compaction, and which may also remove some accumulated thatch. Cores (visible on the surface) are either ground back into the turf as a topdressing or removed and discarded

Once established, a turfgrass is rarely drastically disturbed, and never fully inverted, except at renovation. The most common cultivation techniques are aerification via holes punched in the turf (Fig. 10.2) and underlying soil, or vertical mowing (Fig. 10.3), where the turf is sliced with sharp blades, similar to those of a rotary saw. These cultivation procedures are largely performed for two functions: to lessen thatch accumulation and to reduce compaction of the underlying soil. A unique characteristic of turfgrasses is thatch – a layer of intermingled living and dead plant material that underlies the green turf canopy and overlies the soil (Fig. 10.4). It has been shown that this thatch is a temporary C sink (Raturi et al. 2005) and that N in thatch can greatly affect C:N cycling. For example, in one study that used N^{15} labeled fertilizer, 62% of that N was in Kentucky bluegrass thatch at 18 days after a Fall treatment (DAT), but by 233 DAT the percentage in N in that thatch had reduced to 35%, indicating rapid immobilization of N (Miltner et al. 1996). Total, soluble and lignin C were significantly greater in thatch of both cool (*Agrostis palustris* Huds.) and warm (*Zoysia japonica* Steud.) season turfgrasses, and was significantly greater than that measured in underlying soil (Raturi et al. 2005).

Thus, turfgrasses are a unique management scenario – rarely inverted, a long-term perennial crop, and one in which the harvest (clippings) is performed frequently. In many cases turfgrasses are also intensively managed, with frequent fertilizations, irrigation when drought occurs, and applications of pesticides to maintain turf at some aesthetic standard. Turfgrasses are also different in that they are often part of a landscape system. With the exception of athletic fields or sod farms, turfgrass is some part of a landscape that also includes trees, shrubs, annual and/or perennial plants and hardscapes (sidewalks, driveways, paving).

Fig. 10.3 Vertical mowing of turfgrass, a practice commonly done to reduce thatch, stimulate shoot growth, and smooth the turfgrass surface. If the vertical mowing is severe, accumulated debris on the turf surface will be removed and discarded

Fig. 10.4 A profile view of the debris removed by the cultivation process of vertical mowing. The thicker layer on the right of the picture is from turfgrass that was fertilized with a higher rate of nitrogen (N), as compared to the turf shown in the left hand side of the illustration

10.4 Carbon Flux from Turfgrasses in the Urban Landscape

It is this entire landscape that is often evaluated for C storage, with turfgrass included as a part of the system. A widely cited early study (Falk 1976) examined the energy uses associated with the 1 year management of a suburban lawn in California. Collected biomass data related to the lawn included clipping yield, tree litter, and root and aboveground biomass. All management, including mowing, fertilization, irrigation and raking was recorded, with various published formulae used to measure energy use. Annual net primary production from this mixed grass lawn was 1,020 $g\,m^{-2}$ (4,467.5 kcal m^{-2}), comparable with previously published grassland and cornfield data (Falk 1976). A total of 1,865 kcal m^{-2} per year in labor, gasoline, irrigation and other inputs was required to produce the suburban lawn, of which irrigation and mowing were the two greatest inputs (Falk 1976).

A later study of urban greenspace used two residential blocks in the Chicago area as the study site, with greenspace defined as 'any soil surface are capable of supporting vegetation' (Jo and McPherson 1995). Collected data included foliage, root and total biomass of numerous tree species, calculated using previously published biomass equations. Thirty-two aboveground biomass samples were taken from 12 common species of other herbaceous plants, and total inorganic and organic C was measured in soil in April and September, to a depth of 60 cm. For the turfgrass, (predominate species were Kentucky bluegrass and fescue (*Festuca* spp.)) three parts were evaluated: (1) removed clippings, (2) remaining verdure (called 'stubble'), and, (3) live roots. Annual uptake from live roots and stubble of the turfgrass was calculated from existing lawn and grassland C uptake formulae. Samples of roots and stubble were collected in Nov and Dec 1992, and in Mar, May, July and Sept of 1993; one growing year. Dry weight of these harvested materials was determined, and C content was determined via combustion. Clippings were collected biweekly from 26 lawns, from Oct 1992 through September 1993, except in winter when mowing was not needed. Dry weight of clippings was converted to C values using the same formula-based methods as for stubble and live roots. Other factors in grass maintenance that were evaluated included fertilization, irrigation frequency and duration, and energy use for landscape maintenance (for turfgrass – mowing), data which was then converted to C estimates using existing conversion formulas.

For the two studied blocks, the percentage of the landscape that was turfed was 37.7% and 26.9%. Carbon storage in the grass was found to follow a predictable pattern, with maximum C in summer (July) followed by a decline into fall and winter (Jo and McPherson 1995). Calculated turnover rates for stubble and root C were 2 and 2.9 years, respectively, indicating that about 50% of the stubble carbon and 34% of the total root carbon would be replaced each year. Average C content in grass was 42.7% on a dry weight basis, with no significant difference in C content between live root, stubble or clippings. Over the year there was a net C release from the turfgrass maintenance, with greatest C release coming from mowing (gasoline

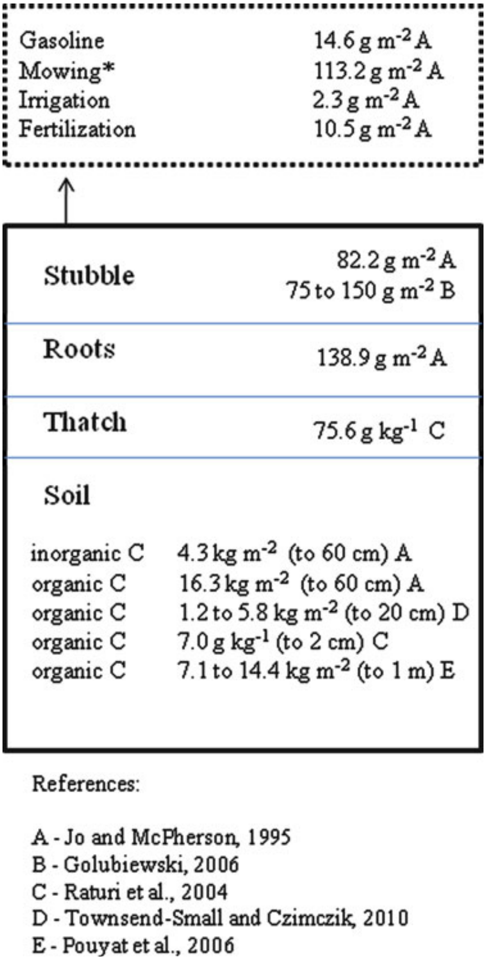

Fig. 10.5 A representative illustration of annual C storage or loss in a turfgrass system. Letters after each number refer to the appropriate reference citation for that number, as referenced at the bottom of the Figure

consumption and C in mown grass) (Fig. 10.5). Total annual C release from mowing, irrigation and fertilization was 0.14 kg m^{-2}, averaged over the two studied blocks. Total C storage averaged 24.7 kg m^{-2} for the two blocks, with soil (0–60 cm) organic C accounting for 83.7% (average of the two blocks) of all stored C. Trees and shrubs accounted for 20.8% to 10.6% of that C (varying with block), while turfgrass accounted for 0.5–0.7%, in both blocks. In this 1 year study, because of mowing, grass returned to the atmosphere 1.5 times the C sequestered. Overall, though, the studied greenspaces were net sinks of C, due to the soils and woody plants (Jo and McPherson 1995).

Others have noted this link between CO_2 emissions from fuel consumption used to manage turfgrass and a lack of organic C sequestration (Townsend-Small and

Czimczik 2010a). In that work, CO_2 emissions from fuel usage of 1,469 g CO_2 m^{-2} $year^{-1}$ were originally reported (Townsend-Small and Czimczik 2010a). The work also shows the dependence on calculations with available formulae, as a correction (Townsend-Small and Czimczik 2010b) revealed that CO_2 emission from fuel usage was actually estimated at 122 g CO_2 m^{-2} $year^{-1}$. Even with this revised emission value, highly managed turfgrass (frequent mowing, fertilization, irrigation) had little organic C sequestration. However, the revised calculations did indicate a potential for urban ornamental lawns to sequester atmospheric CO_2, but only if managed conservatively (Townsend-Small and Czimczik 2010b).

10.5 Quantifying Carbon Sequestration in the Turfed Landscape

The papers discussed above attempted to quantify energy and/or C in soil and turfgrass with the energy required to maintain that turfgrass. A greater number of published papers have simply examined C of soils in the turfgrass or urban landscape, examining changes as affected by climate, region or age of the system. Data collected from turfed landscapes is sometimes compared to other management systems such as crop production, native grasslands, or urban forests.

In Colorado, C fluxes in Kentucky bluegrass lawns were compared to those in irrigated corn (*Zea maize* L.), wheat (*Triticum aestivum* L.)-fallow, and native grasslands (Kaye et al. 2005). In this 1 year study total, inorganic and organic soil C contents were determined in a June sampling, and potential C mineralization was measured using a laboratory incubation procedure. Soil respiration was measured approximately twice a month, and C and N content of all plant tissues was also measured throughout the study. Specifically for the turfgrass, harvested clipping samples were analyzed for N, while stubble biomass turnover was estimated using previously published Kentucky bluegrass data (Falk 1976, 1980; Jo and McPherson 1995). In this study clippings were returned to the lawn, as was common for the area. Compared to the other studied land-uses, total belowground C allocation in the urban lawn system was 2.5–5 times greater (2,602 g C m^{-2} $year^{-1}$)(Kaye et al. 2005). Seasonal variation in C from grass clippings was similar to that observed in Jo and McPherson (1995), with a high in May (43 g C m^{-2} $month^{-1}$), and a steady decline thereafter. The authors concluded that urbanization of arid and semiarid ecosystems led to enhanced C cycling, when compared to native grasslands and agricultural ecosystems (Kaye et al. 2005).

Additional C research in the arid Colorado region studied C pools in lawns, woody vegetation, and soils (Golubiewski 2006). In this 1 year study clippings from Kentucky bluegrass, creeping red fescue (*Festuca rubra* L.), buffalograss (*Buchloe dactyloides* L.) or mixed grass lawns was harvested twice a month from April through October, with verdure (stubble) biomass harvested at the end of the growing season. Both plant tissue and stubble biomass C were determined via dry combustion.

Root or other belowground tissue C was not measured. Lawns varied in age, with approximately 10 lawns each dating from 1900 to 1940, 1950s, 1960s, 1970s, and 1980s–1990s. Thirty nine lawns were sampled for tissue N and C, and 14 lawns had soil (to 30 cm) sampled for determination of total, organic and inorganic C.

Carbon concentration in the harvested clippings was relatively similar over harvests and species, ranging from 42.8% to 45.7% (Golubiewski 2006). Carbon content in the lawn system was compared to that of grasslands and agricultural fields, with the grassland and agricultural crop C data gleaned from previously published work. Newly constructed lawns (10 year old or younger) stored less soil and stubble C than did older (1970s and older) lawns, and every lawn system stored more C than cultivated (conventional tillage) systems. Lawns and underlying soils that were at least 20 years old had greater C than grasslands (Golubiewski 2006). The authors concluded that, after an initial decrease in C pools due to landscape construction, mature and older lawns had greater soil and stubble C than grasslands. However, it was noted that yard maintenance and fertilization could offset C storage via C emissions, and that additional C budget research is needed (Golubiewski 2006).

The soil organic carbon data collected by Golubiewski (2006) was used again in a 2009 study, in which soils in the Baltimore, MD area were sampled from both residential lawns and suburban remnant forest (Pouyat et al. 2006, 2009). Unlike the CO data set, in which lawn age spanned a wide age range, lawns in one part of the Baltimore study largely dated from before 1950, and all forest remnants were at least 80 years old. A second set of Baltimore lawns dated from the 1970s and 1980s. For the 2009 study only soil C was studied - no clipping or remaining lawn stubble data was collected. Soil organic C values reported in the paper ranged from 5.45 kg m^{-2} for cultivated soils in Baltimore to 8.47 for urban lawn soils in Denver (Pouyat et al. 2006; Golubiewski 2006). Highest soil organic carbon values were for the > 80 year old remnant forests in Baltimore, with average soil organic carbon of 11.0 kg m-2 (Pouyat et al. 2009). Comparisons of soil organic C between Baltimore and Denver indicated that residential lawns had the capability to accumulate similar levels of soil organic C, regardless of regional differences in climate, parent material and topography (Pouyat et al. 2009).

A recent study evaluated C sequestration amongst different turfgrass species, the first study in which turf species was a treatment (Qian et al. 2010). The cool season grasses fine fescue (both irrigated and non-irrigated), Kentucky bluegrass (irrigated), and creeping bentgrass (irrigated) were evaluated for changes in soil organic C, soil C sequestration and soil organic C decomposition. At 4 years after establishment approximately 17% to 24% of soil organic C (0–10 cm depth) was contributed by the established turfgrass. The amount of soil organic carbon differed with turfgrass species and irrigation, with irrigated fine fescue having the highest soil organic C (3.4 Mg C ha^{-1} $year^{-1}$). Every turfgrass species sequestered C during the first 4 years after turf establishment, with highest rates for irrigated fine fescue and creeping bentgrass (Qian et al. 2010).

One of the few published turfgrass carbon studies that utilized applied treatments, a 2008 study measured carbon dioxide emissions from four different levels of lawn management (Allaire et al. 2008). The four lawn management approaches used on the mixed cool-season lawn were: (1) N fertilizer with clipping removal and frequent mowing, (2) no fertilizer with clippings returned and frequent mowing, (3) treatment #2 with only 3 mowings, and, (4) treatment #2 with only 1 mowing. Treatments were imposed for 1 year, after which weekly CO_2 emission data was collected. Mowing frequency had a greater impact on CO_2 flux than did fertilization. Frequently mowed lawns had an annual CO_2 emission of 2.0 kg m^{-2} $year^{-1}$, which was four times higher than in lawns mowed 3 or 1 times. At a high mowing frequency clipping removal did not affect CO_2 emissions (Allaire et al. 2008).

10.6 Modeling C in Turfed Landscapes

Long-term soil test data and the CENTURY model were utilized in a series of papers that assessed soil C sequestration in golf courses located in the Denver and Fort Collins, CO area (Qian and Follett 2002). Courses ranged in age from 2 to 45 years, with prior use mixed between native grassland and agriculture. Using nonlinear regression analysis it was determined that the time for soil organic matter to reach equilibrium for fairways and putting greens was 31 and 45 years, respectively. Total C sequestration in fairways during the first 25 to 30 years after establishment was estimated at 0.9 t ha-1 $year^{-1}$, with the most rapid C sequestration estimated in that first 25–30 year period (Qian and Follett 2002). Because it is so costly and time consuming to collect soil C data, the CENTURY model was used to model the long-term effects of clipping and N management on soil organic carbon (Bandaranayake et al. 2003; Qian et al. 2003). Clippings of Kentucky bluegrass collected under home lawn conditions were used to verify the model, and the model was used to predict biomass production and soil organic C and N. When clippings were returned for 10–50 years to the lawn the model predicted that soil C sequestration would be increased by 11–25% (Qian et al. 2003). When the golf course soils data (Qian and Follett 2002) was used in the CENTURY model results indicated that the turfgrass was serving as a C sink, with 23–32 Mg ha^{-1} of soil organic C sequestered after 30 years (Bandaranayake et al. 2003).

Accumulations of soil C in long-term stands of turfgrass was studied in a series of papers which used a common data set of soils sampled from golf courses in the Pinehurst, N.C. area. The hybrid bermudagrass courses, which had similar sand/loamy sand soils, were 95, 23, 6 and 1 years old at the time of the study (Shi et al. 2006b, c). Long-term undisturbed soil planted with native pine trees (*Pinus palustris* Miller) was also sampled. Collected data included total soil C, organic C and soil microbial biomass C (Shi et al. 2006b). Soil C decomposition and N mineralization/immobilization as affected by clipping addition was evaluated in a 28 day incubation (Shi et al. 2006a). Decomposition of the soil organic C, and the soil enzymes that regulate that decomposition were also studied (Shi et al. 2006c). Last, the Pinehurst data set was compared to a data set created from the sampling of three Las Vegas, NV golf courses which were 6, 19 and 44 years old at sampling. The objectives of that study were to examine N fertilization, soil pH and clipping effects on enzyme activities and soil organic C (Yao et al. 2009).

As the NC golf course fairways aged total soil C (both 0–5 and 5–15 cm sampling depths) increased, with significantly more soil C measured in the 95 year old soils. Soil sampled from native pines had significantly more soil C (26.2 mg C g-1) than soils from the 6 year old golf course (9.5 mg C g-1). These soil C levels were significantly less than that measured in soils from the 23 (38.4 mg C g^{-1}) and 95 year old (72.5 mg C g^{-1}) soils (Shi et al. 2006b).

Another study that examined soil C in turfgrass soils of differing ages did so in New Zealand putting greens that were 5, 9, 20, 30 and 40 years old (Huh et al. 2008). Soils were sampled at 5 depths to a final depth of 0.25 m, with soil analyzed for total and organic carbon. Aboveground plant tiller production was also measured for plant-biomass production, and it was found that tiller numbers did not change with putting green age, indicating that above-ground biomass production was similar across putting green age. Soil C sequestration in the 0–0.25 m depth of the putting green averaged 69 g m^{-2} $year^{-1}$, and this soil C increased linearly as the greens aged. Soil organic C in the 5 year old green was 1.5 kg m^{-2} (0–0.25 m depth), and was 4.0 kg m^{-2} in the 40 year old green (Huh et al. 2008).

Studies which evaluate C sequestration in urban settings in the southeastern United States are somewhat absent from the refereed literature. Most published work focused on arid/semiarid Colorado, or established areas in Denver and Baltimore. One study utilized remotely sensed data to estimate the regional impacts of land use on Net Primary Productivity (NPP). In this work turfgrass was combined with trees and other urban vegetation. Other land cover classes included categories such as crops, evergreen forest, and grassland (Milesi et al. 2003). Mean NPP of urban land cover across the seven studies SE states ranged from a low of 749 g C m^{-2} $year^{-1}$ in FL to a high of 848 g C m^{-2} $year^{-1}$ in GA. The higher mean NPP estimated in GA was hypothesized to be a result of the presence of parks and golf courses. Overall, increased urbanization of the SE (from 1992 to 2000) resulted in a 0.04% reduction in estimated annual NPP (Milesi et al. 2003). In another remote sensing study, areas of turfgrass (estimated at 163,800 km^2 (1.9% of the surface area) in the continental United States) were compared to impervious surface areas (Milesi et al. 2005). Modeling of turfgrass growth indicated that a well maintained cool- or warm-season lawn was a C sequestering system, although the authors noted that the use of large amounts of water and N would affect this negatively (Milesi et al. 2005).

10.7 Summary and Future Needs

A review of the published refereed research in C sequestration/flux in turfgrass systems indicates the following:

- As turfgrass ecosystems aged, sequestration of soil C increased linearly with age. These increases were observed in some soils that had been turfed for up to 95 years.

- Much of the research which focuses on C flux or sequestration in urban lawns does so in cool -season turfgrass management systems. With the exception of work from NC, warm-season turfgrasses are largely unstudied.
- Almost all the of the work evaluates some form(s) of soil C, and work which analyzes collected turfgrass clippings, shoots or stubble for C is scant. Information on the carbon content of turfgrass roots and clippings is needed, as well as data on the impact of the specialized turfgrass stems (stolons and rhizomes) on soil C. This tissue work is needed across the many diverse turfgrass species.
- Data on the sequestration of C in turfed soils often utilized the one-time sampling of sites with a documented turf age and management program, so that age of the turfgrass system was the treatment. Studies that imposed a set of treatment effects and then evaluated those effects on C are (somewhat logically), much more limited.
- In whole system ecological studies where biomass or turnover data was estimated from previously published work, a few key papers were often frequently and repeatedly cited (Falk 1976, 1980). Calibration data, especially for use in models, is lacking.
- Almost every published study is 1 year of sampling. Multi-year studies, especially those in which treatment effects such as irrigation, species selection, or N fertilization are imposed, are greatly needed.
- The impact of turfgrass management practices, especially mowing, N fertilization, and irrigation, on C sequestration is the area which needs greatest study. The few studies which attempted to quantify the impact of turfgrass management present preliminary evidence that intensive mowing, irrigation and fertilization may eliminate the C sequestration benefits of turfgrass. Data is quite limited, however, and this area deserves additional study.

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Chapter 11
Microbial Control of Soil Carbon Accumulation in Turfgrass Systems

Wei Shi, Daniel Bowman, and Thomas Rufty

Abstract Turfgrass is a major component of urban and suburban landscapes. It appears that large amounts of carbon (C) can be sequestered in turfgrass systems, making them, potentially, important contributors to CO_2 mitigation efforts. The chronosequences in the present study and those published by others indicate that soil organic matter (SOM) accumulation beneath turfgrasses is non-linear, progressively moving towards stability over time, which is a pattern typical of restored ecological systems. With this hyperbolic accumulation, SOM and C accumulation rates steadily decline as turfgrass systems age, and rates approach minima after 25–40 years. Because turfgrass growth and generation of C generally do not decline over time, decreasing accumulation rates must result from increased SOM degradation by soil microbes. The size of soil microbial population and its activity increase with the accumulation of SOM; the highest activity being near the soil surface. Examinations of microbial diversity using fatty acid methyl ester and community-level physiological profiling have shown that a diverse community is present soon after turfgrass systems are established and the community is sustained, with little change, for periods of up to 100 years. Little impact from continual use of fertilizers and pesticides can be detected. Soil enzymes are abundant in turfgrass systems. Hydrolytic and oxidative enzyme activities contribute to the microbial degradation process.

Keywords Microbial activity • Soil enzyme • Soil organic matter • Soil C sequestration • Turfgrass

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List of Abbreviations

CLPP	community-level physiological profiling
DOC	dissolved organic C
MBC	microbial biomass C
MBN	microbial biomass N
Mha	million hectares
FAME	Fatty acid methyl ester
SOC	soil organic C
SOM	soil organic matter
SON	soil organic N

11.1 Introduction

As detailed in other Chaps. (1, 2, 3), turfgrasses are an important part of urban and suburban landscapes. Their use as roadsides, residential lawns, parks, sport fields, and golf courses totals ~16.4 million hectares (Mha), making turf the single largest irrigated crop in the U.S. (Milesi et al. 2005). Concerns about climate change continue to mount, especially considering recent analyses that emphasize the compelling seriousness of the threat (Hansen et al. 2008; Allen et al. 2009a; Schneider 2009; Solomon et al. 2009). Thus, it is readily understandable that there is an intense interest in CO_2 mitigation (Allen et al. 2009b; Lal and Follett 2009; Washington et al. 2009) and C sequestration in turfgrass systems (Qian and Follett 2002; Qian et al. 2003, 2010).

This chapter explores the role of soil microbial activity in the sequestration of C in turfgrass systems. Net accumulation of C in soil reflects the balance between C-compounds generation by plants and subsequent degradation by microbes. To borrow from Jastrow et al. (2007), net accumulation reflects 'disequilibrium' between the two processes, ultimately favoring generation. This chapter discusses that the balance between generation and degradation evidently changes with time, and the long-term pattern of C accumulation may be largely determined by the activity of a dynamic soil microbial population. The case is first made by examining long-term kinetics of C accumulation revealed in chronosequences. The chapter then focuses on the types of changes observed in microbial communities and activities that appear to be involved in regulation of soil C accumulation in the turfgrass system.

11.2 Conceptual Framework

Turfgrasses are routinely established during development of urban and suburban landscapes. Most often, they involve restoration of a perturbed soil system, and offer a model ecosystem in which to study soil organic matter (SOM) accumulation.

Surprisingly, few studies have examined C sequestration in turfgrass systems, but most all have shown that soil C can accumulate at relatively high rates, which have ranged from 64 to 174 g C m^{-2} $year^{-1}$ (Table 11.1). As a comparison, average rates of C accumulation following transition from abandoned agricultural fields to forests and grasslands were estimated at ~33 g m^{-2} $year^{-1}$ (Post and Kwon 2000).

Some caution is warranted before accepting a specific rate for soil C accumulation. Restored ecological systems typically accumulate C rapidly in early years, but then slow to much lower rates (refer to Schlesinger 1990), i.e., rates are highly dependent on time. The pattern of progressively slowing accumulation is typical for natural and agricultural systems, as soil C steadily moves towards equilibrium (Odum 1969; Vitousek and Reiners 1975).

A recently completed study of soil C accumulation beneath fairways on golf courses in North Carolina (See Image 11.A) provides a relevant example. The kinetic pattern of long-term C accumulation is revealed in the assembled 'chronosequence' (Fig. 11.1). Soil organic C (SOC) is quantified with and without identifiable root material at a 0–15 cm depth (See Image 11.B). The golf courses were located in Coastal Plain and Piedmont regions and ranged in age from 2 to nearly 100 years. Although there were differences in soil types, most North Carolina soils are classified as Ultisols (Buol et al. 2003), which are highly weathered and acidic, and typically have low levels of SOM concentration prior to turfgrass establishment. All fairways were planted with monostands of bermudagrass (*Cynodon dactylon* (L.) Pers. or *C. dactylon* x *transvaalensis* Burt-Davy), providing a relatively uniform ecosystem compared to natural vegetation stands or crop lands under rotation.

Data from the chronosequence show that soil C accumulates rapidly below in young turfgrass stands, but then tends to level off with age (Fig. 11.1a). The pattern is best described by a hyperbolic equation, which has an inherent decline in the rate of net accumulation. Based on the slope (first derivative) of the hyperbolic accumulation curves (Fig. 11.1b), the rate of soil C accumulation in the bermudagrass system declines from 6 Mg ha^{-1} $year^{-1}$ to much less than 0.5 Mg ha^{-1} $year^{-1}$ over the first 20 years.

Theoretically, one might assume that net accumulation of soil C during the early years could provide a reasonable estimate of the potential for generation of soil C in the system over the longer term. Using the chronosequence without root material as a model, the rate in the initial 3 years was ~ 5 Mg ha^{-1} $year^{-1}$ (see dashed line, Fig. 11.2). This is much higher than the actual rate of C accumulation over an extended time period.

Why does the rate of C accumulation decline over time? From the measurements in the field and from anecdotal observations from turfgrass managers and agronomists, there is no indication that growth of bermudagrass and C production slow over time. Assuming this to be true, the pattern implies intense degradation of SOM by microbial activity as the system ages. The pool of C that accumulates in soil, as shown in Fig. 11.2, can be viewed as 'protected' and resistant to microbial degradation.

Table 11.1 Soil organic carbon accumulation in turfgrass systems

Turf use	Location	Turf age (year)	Soil depth (cm)	SOC accumulation rate[a] (Mg C ha^{-1} year^{-1})	SOC content[b] (%)	References
Golf course fairway	Colorado, USA	1.5–45	0–11.4	~1	1.0–2.4	Qian and Follett (2002)
Golf course putting green	Colorado, USA	1.5–45	0–11.4	~1	0.4–2.3	Qian and Follett (2002)
Golf course fairway	Wyoming, USA	18–34	0–15.2	~1	0.7–1.4	Qian and Follett (2002)
Golf course putting green	Palmerston North, New Zealand	5–40	0–25	~0.7	0.1–1.1	Huh et al. (2008)
Golf course putting green	North Carolina, USA	1–25	0–15	~0.6		Seth-Carley et al. (2011)
Lawn	California, USA	2–33	0–20	~1.4	0.4–1.9	Townsend-Small and Czimczik (2010)
Golf course fairway	North Carolina, USA	5–99	0–7.5	~1.4	0.7–4.6	Yao and Shi (2010)
Golf course fairway	Nevada, USA	6–44	0–7.5	~1.7	3.2–5.1	Yao and Shi (2010)

[a] Soil bulk density 1.5 g cm^{-3} is used for converting soil organic carbon content based on soil mass

[b] Soil organic carbon content ranges from the lowest to the highest values in the turfgrass systems of different ages

Image 11.A A golf course fairway like those sampled for the turf chronosequences

How solid is the evidence that rates of C accumulation in soil *decrease* over time in turfgrass systems? Although a limited number of studies from chronosequences have been published, most present similar patterns of C accumulation. Working in Colorado, Qian and Follett (2002) reported that C accumulation in bentgrass greens and fairways was non-linear with declining accumulation rates over several decades. Similarly, chronosequences of bentgrass putting greens in North Carolina (Seth-Carley et al. 2011), as well as simulations of C and nitrogen (N) accumulation in soils of fairways and home lawns (Century Model-Bandaranayake et al. 2003; Qian et al. 2003), consistently indicated falling accumulation rates. Declining accumulation rates can also be inferred from decreasing soil N accumulation rates in turfgrass systems in the Northeast (Porter et al. 1980; Petrovic 1990), assuming that C:N ratios remained relatively stable. As with the data from the present study for bermudagrass (Fig. 11.1), the C accumulation patterns in the various published studies, collectively, indicate a plateau with low accumulation rates occurring after about 25–40 years.

Two published studies indicate that C accumulation beneath turfgrasses is linear (Huh et al. 2008; Townsend-Small and Czimczik 2010). Evaluation of these is difficult, because they involved samples from a relatively narrow chronosequence. Obviously, SOM cannot accumulate without limit; describing accumulation as constant is thus a matter of selecting a “short” time frame and curve fitting the data.

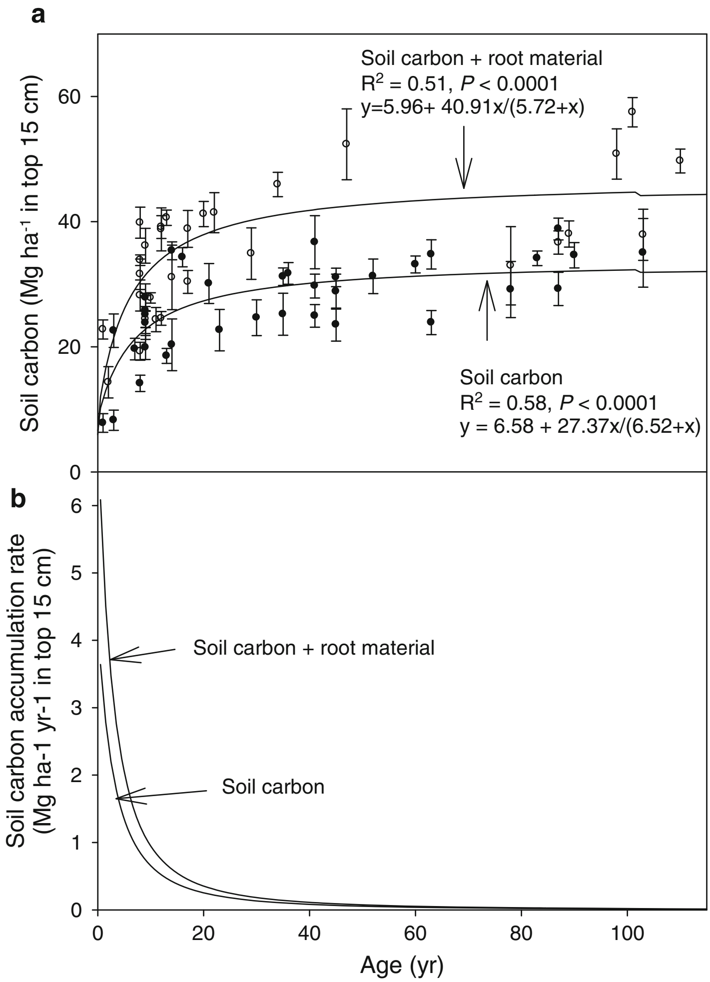

Fig. 11.1 Soil organic carbon accumulation (**a**) and accumulation rate (**b**) as a function of age of bermudagrass turf in North Carolina, USA. Soil organic carbon is quantified with and without root material. Samples were taken at a depth of 0–15 cm

11.3 The Relationship Between SOM and Microbial Activity

Conceptually, a decreasing rate of net C accumulation largely controlled by increasing degradation necessitates the involvement of a dynamic, active microbial population. It is generally accepted that soil microbial population size increases with accumulation of SOM (Jenkinson and Ladd 1981), even under unfavorable soil conditions (Insam and Domsch 1988). Indeed, microbial biomass C (MBC) and N (MBN) and

Image 11.B A soil core of the top 15 cm from a bermudagrass fairway that is 60 years

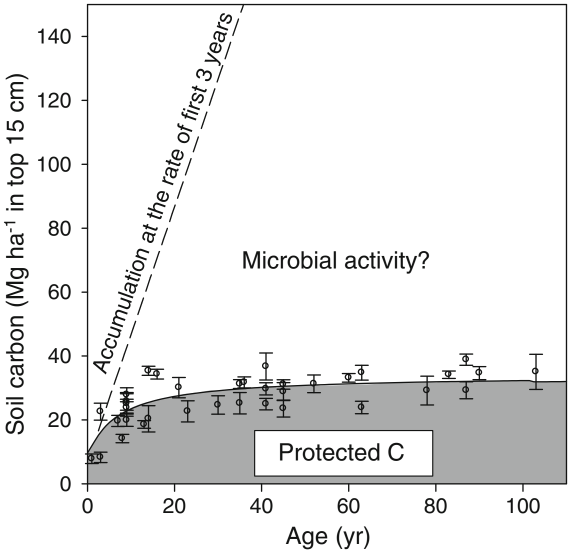

Fig. 11.2 Comparison of the actual pattern of soil carbon accumulation with that extrapolated from the initial rate of accumulation (3 years) for bermudagrass turf in North Carolina, USA

Table 11.2 Soil organic matter and microbial biomass in turfgrass systems of different ages

	Soil C	Soil N	MBC	MBN	MBC: Soil C	MBN: Soil N
	mg C or N g^{-1} soil		μg C or N g^{-1} soil		%	
0–5 cm depth						
Native pines	26.2c	0.9d	217.2c	30.8c	0.84c	3.44c
1-year turf	9.5d	0.6d	180.0c	40.0c	1.91a	6.58a
6-year turf	30.4bc	2.3c	564.1b	102.0b	1.86ab	4.47b
23-year turf	38.6b	3.1b	663.0b	112.5b	1.77ab	3.77bc
95-year turf	72.5a	7.0a	1125.9a	219.3a	1.56b	3.17c
5–15 cm depth						
Native pines	9.5b	0.4bc	68.4b	11.2b	0.73c	3.23a
1-year turf	2.3c	0.2d	25.6c	4.8d	1.13a	2.41ab
6-year turf	2.8c	0.3 cd	28.2c	5.4d	1.04ab	1.79b
23-year turf	8.2b	0.5b	53.4b	9.1c	0.65c	2.02b
95-year turf	13.3a	1.1a	109.0a	21.8a	0.83bc	2.03b

Different letters within each column of 0–5 or 5–15 cm soil depth indicate the significant difference of mean value ($P < 0.05$) by multiple comparisons of Bonferroni *t*-test (Compiled from Shi et al. 2006a)

microbial activity were found to be positively correlated with the accumulation of SOM in aging turfgrass systems (Table 11.2; Shi et al. 2006a, 2007; Yao and Shi 2010).

The SOM accumulation in turfgrass soils decreases with soil depth. The most rapid accumulation in bentgrass greens, for example, is near the soil surface (Carrow 2003, 2004), coincident with the largest microbial populations and high microbial activity per unit of C (Mancino et al. 1993; Bigelow et al. 2002; Huh et al. 2008). The corresponding high rates of degradation near the soil surface are in contrast to those deeper in the soil profile where fewer roots occur and SOM contents are lower. In that zone, slow C accumulation may appear linear over a relatively long period (Huh et al. 2008; Seth-Carley et al. 2011).

Disequilibrium between SOM generation and decay may also involve different sub-populations within the soil microbial community. One functional component of the microbial community responded rapidly to the addition of grass clippings and was independent of indigenous SOC and microbial biomass (Shi et al. 2006b; Yao et al. 2009). This microbial fraction resembled r-strategy organisms that count on high reproductive rates for survival and fluctuate with available C and nutrient resources (Bottomley 2005). In turfgrasses, r-strategy organisms would be decomposing organic compounds contained in fresh plant materials, such as the grass clippings, fine roots, and root sloughs and exudates which would be present in the largest quantities in the zone of intense growth near the soil surface. By contrast, microbes that do not respond to the addition of grass clippings may play a significant role in decomposing more stable SOM and could be labeled as k-strategy organisms, which are associated with the breakdown of indigenous SOM.

Even though the overall pattern of soil C accumulation indicates more rapid degradation as turf systems age, a fraction of SOM remains ‘protected’ and resistant

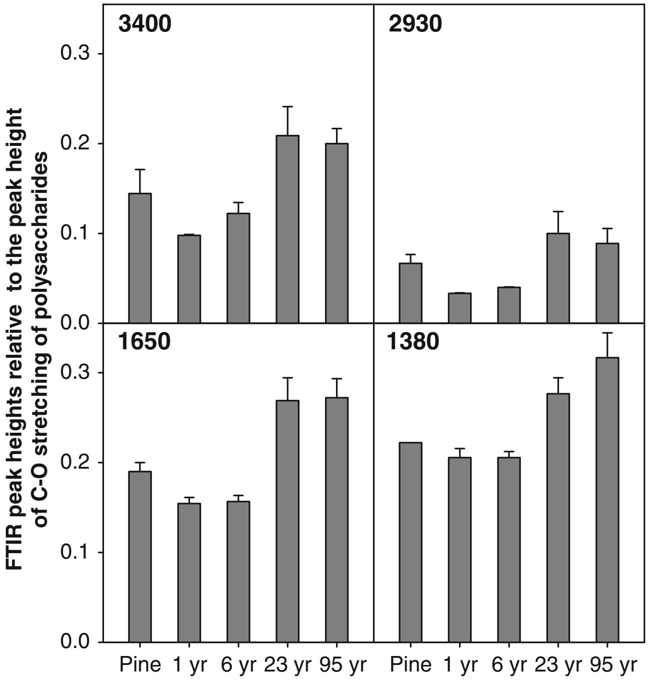

Fig. 11.3 Ratio of FTIR (Fourier transform infrared spectroscopy) peak height at 3,400, 2,930, 1,650, and 1,380 cm^{-1} to the peak height of the C-O stretching of polysaccharides for soil organic matter (0–5 cm soil depth) collected from bermudagrass fairways of different ages and adjacent native pines. Different letters within each bar group of individual soil depth indicate significant differences of mean values at $P<0.05$ (Compiled from Shi et al. 2006b)

to degradation (refer to Fig. 11.2). A conceptual model has been proposed to account for the various constraints on microbial activities that result in soil C stabilization (Six et al. 2002, 2006; Jastrow et al. 2007). Physically, SOM can be occluded within soil aggregates; chemically, SOM can be adsorbed to soil particles (i.e., silt and clay minerals), both restricting access by soil enzymes and microbes.

The protection of SOM, together with microbial preferential use of easily decomposed organic materials, leads to increases in the relative abundance of recalcitrant SOM over time (Shi et al. 2006b). A series of observations support this view. Analyses of chemical functional groups within SOM using Fourier transform infrared spectroscopy showed that aliphatic and carboxylic compounds, including those with O-H and N-H stretching, increased with turf age (Fig. 11.3). Higher ratios of peaks representing aliphatic, phenols, and carboxylic acids were found in soil from older turfgrass systems. The ratios of mineralization to total SOC and N declined significantly with age (Shi et al. 2006a). And the ratio of soil dehydrogenase activity to SOC concentration declined 50% over a 40-year period (Huh et al. 2008).

Recent experiments examined whether recalcitrance in the form of chemical resistance was produced by microbial processing (i.e., decomposition and resynthesis) or simple preservation of organic compounds of plant materials (Yao and Shi 2010). The natural abundance of ^{13}C and ^{15}N in total SOM was measured along with the light and heavy fractions from turfgrass soils. Soil microorganisms preferentially metabolize the light isotopes ^{12}C and ^{14}N, enriching the SOM with the heavy isotopes ^{13}C and ^{15}N. With increased turfgrass system age, the abundance of ^{13}C and ^{15}N in SOM increased, suggesting an association with microbial processes. It had been proposed previously that the microbial contribution to soil C storage was related directly to microbial community dynamics and the balance between formation and degradation of microbial byproducts (Six et al. 2006).

11.4 Impacts of Management Practices

Ecosystem models have been used to predict changes in SOM accumulation with differing inputs. The CENTURY model which simulates C and nutrient dynamics in plant-soil systems (Qian et al. 2003) and the Biome-BDG ecosystem process model (Milesi et al. 2005) both predict that increasing resource inputs (i.e., fertilization, clipping return, and irrigation) will increase net SOM accumulation. Consistent with the model predictions, irrigation was found to increase primary productivity, organic C input, and decomposition of turfgrass clippings in a recent study (Qian et al. 2010). Additionally, high resource availability seemed to offset climatic differences in a study comparing soil C beneath turfgrasses in the eastern (Baltimore, MD) and western (Denver) U.S., where smaller than expected variations in soil C accumulation occurred (Pouyat et al. 2009).

11.5 Soil Microbial Community Structure

Often, managed ecosystems have less diverse microbial communities compared to their natural counterparts (Torsvik et al. 2002). In cultivated soils, the decline in diversity is probably related to reductions in SOC concentration (Degens et al. 2000). In turfgrass soils, however, SOM accumulates to relatively high concentrations (Qian and Follett 2002; Bandaranayake et al. 2003; Yao and Shi 2010), which should promote diversity. Conversely, a number of turf management practices might counteract the positive impacts of SOM. Construction activities during new turf establishment change plant species and soil properties, disrupting the ecology (Nüsslein and Tiedje 1999); turfgrasses are usually planted and managed as a monostand; and turfgrass management involves frequent additions of fertilizers and pesticides that could depress the microbial community over time. Intense management has been found to cause shifts in soil microbial community composition and structure in agricultural soils and managed grasslands (Ka et al. 1995; Lundquist et al. 1999; Donnison et al. 2000; Webster et al. 2002; Clegg et al. 2003).

While it is known that multiple agronomic variables can affect composition of the soil microbial community, their relative importance is not clear. In one study, soil tillage and chemical use were the primary factors affecting soil microbial community composition and structure compared to plant species (Buckley and Schmidt 2001). In another, soil type was most important in structuring the soil microbial community compared to factors such as seasonal change, specific farming operation, management system, and spatial variation (Bossio et al. 1998). Soil type was singled out as the primary determinant of soil microbial community structure in arable soils as opposed to cropping system (Girvan et al. 2003), providing some observational consistency.

Over the past several years, authors of this chapter have examined microbial community structure in soil from turfgrass systems. Soils from a turfgrass chronosequence extending from 1 to 95 years of age were examined using fatty acid methyl ester (FAME) and community-level physiological profiling (CLPP) techniques (Yao et al. 2006). Throughout the chronosequence, bermudagrass fairways had been established from pines, the native vegetation in the area. Both CLPP and FAME-based principal component analyses revealed that there was considerable microbial biodiversity, with distinct groups of soil microbial communities present in soils across the turfgrass chronosequence. The communities differed from those in adjacent pine stands. The CLPP analysis also showed that the soil microbial community in turfgrass soils preferred carbohydrates while those in native pine soils preferentially utilized phenolic compounds and carboxylic acids. The difference in catabolic functions was mirrored by a compositional change of FAME. The fatty acids a15:0, 16:1, 16:0, 18:1 and cy17:0 were positively correlated with the soil microbial community in turf soils, while the fatty acids cy19:0 and i16:0 were correlated with the microbial community in native pines. Cluster analysis could detect only a small degree of divergence in the soil microbial communities in young and old turfgrass systems, once soil data from native pine were removed from the data sets. The results from the studies have clearly indicated that a diverse soil microbial community was established and sustained in turfgrass soils. The major shift in soil microbial community structure occurred at the initial land use change. A similar conclusion was reached in a study of turfgrass soils in different geographic locations where similar microbial community compositions were examined using FAME (Bartlett et al. 2007). Thus, it seems evident that turf management practices, such as continual applications of fertilizers and pesticides, have only minor impacts on the structure of the microbial community.

It should be pointed out that the SOM/diversity relationship seen in the turfgrass chronosequence is not entirely stable. Spatial examinations of the soil profile revealed that microbial diversity was noticeably lower at a greater depth (5–15 cm) in the two youngest turfgrass systems (Yao et al. 2006). The SOC concentrations were lower in that zone, less than 3 $g\,C\ kg^{-1}$. As the systems aged and SOM accumulated to higher levels, soils in that zone had nearly constant FAME-based Shannon-diversity indices as C increased from 8 to 72 $g\,C\ kg^{-1}$. Thus, even though some discontinuity exists in the SOM/microbial diversity relationship, it seems that the high levels of SOM accumulating beneath turfgrasses are the primary determinant of the high microbial diversity.

There is some evidence indicating that soils beneath turfgrasses are dominated by bacteria, with a lesser role for fungi. This possibility is based on fatty acid signatures, along with microbial C:N ratios in the 4–6 range which are common for bacterial populations (Shi et al. 2006a; Yao et al. 2006). It has been suggested that a dominant fungal community will favor soil C storage (Six et al. 2006). Logically, this leads one to think that bacterial dominance is an important factor for the high rates of SOM degradation in turfgrass systems.

11.6 Turfgrass Soil Biochemistry

As argued above, SOM degradation must be the dominant regulator of C sequestration in turfgrass soils over extended periods of time. Multiple mechanisms at different scales are responsible for the high rates of degradation. In addition to the microbial population size and its activity, degradation of SOM involves a suite of soil enzymes produced by soil microbes that catalyze diverse biochemical reactions. It is generally accepted that enzymatic depolymerization is a rate limiting step of decomposition. While hundreds of extracellular enzymes may exist in a soil, a few have been linked closely with soil C and nutrient dynamics – soil cellulase, chitinase, and ligninase.

11.6.1 Hydrolytic Enzymes

Cellulase is a group of hydrolytic enzymes that catalyze the breakdown of glycosidic bonds in cellulose. Complete degradation of cellulose requires at least three enzymes: endo-β-1,4-glucanase, exo-β-1,4-glucanase, and β-glucosidase (Alef and Nannipieri 1995). Chitinase is an essential class of enzymes that hydrolyzes glycosidic bonds in chitin. As the major component of insect exoskeletons and fungal cell walls, chitin is a significant component of the soil organic N (SON) pool (Paul and Clark 1996). Three enzymes, chitinase, chitobiase, and N-acetyl-β-glucosaminidase act in consort for the complete degradation of chitin. N-acetyl-β-glucosaminidase is often used as the indicator for the activity of soil chitinase.

Few studies have examined soil enzyme activity in relation to SOM in turf systems. Yao et al. (2009) characterized soil hydrolytic enzyme activities from two turfgrass chronosequences, one with turfgrass growing on an acidic soil and the other on a calcareous soil. The SOM accumulated slower in the acid soil conditions. Activities of soil cellulase, glucosidase, chitinase, and glucosaminidase were correlated with amounts of SOM accumulating in both chronosequences, and addition of grass clippings significantly increased their activities in both. However, hydrolytic enzyme activities were lower in the calcareous soils where SOM accumulated most rapidly.

Nitrogen fertilization may stimulate the production and activity of cellulolytic enzymes (Fog 1988; Berg and Matzner 1997), thereby reducing SOC storage in forest soils (Waldrop et al. 2004; Sinsabaugh et al. 2005). However, hydrolytic enzymes activities were little affected by mineral N input in turf (Yao et al. 2009).

11.6.2 Oxidative Enzymes

Ligninase represents a group of oxidative enzymes that modify phenolic-containing organics including polyphenols, lignin, and humus. As a typical ligninase, soil phenol oxidase acts upon phenolics of various complexities by catalyzing the release of oxygen radicals (Hammel 1997; Claus 2004). Through this modification, simple phenolics may be fully degraded, and complicated ones partially oxidized with generation of phenolic intermediates (Burke and Cairney 2002; Claus 2004; Toberman et al. 2008).

Soil phenol oxidase regulates SOM decomposition and CO_2 efflux in peatlands and some upland ecosystems (Freeman et al. 2001a; Sinsabaugh et al. 2008). In direct effects, phenol oxidase acts upon otherwise recalcitrant compounds and stimulates decomposition. Indirectly, phenol oxidase enhances decomposition by degrading soluble phenolic compounds and thus eliminating their inhibition of hydrolytic enzyme activities. This indirect pathway has been identified as an 'enzymatic latch mechanism' that underlies soil C loss from drained peatlands (Freeman et al. 2001a).

Phenol oxidase activity was relatively high in an alkaline turf soil, where it appeared to enhance phenolic solubility and in turn inhibit the activity of soil hydrolases such as cellulase, glucosidase, and chitinase (Yao et al. 2009). Similar results have been reported in other ecosystems (Freeman et al. 2001b; Fenner et al. 2005; Nambu et al. 2008). As a consequence of this 'enzymatic latch mechanism' in alkaline soils, one might expect that decomposition of complex organic compounds would be incomplete in those turfgrass systems, leading to accumulation of low molecular weight compounds, i.e., dissolved organic C (DOC).

11.6.3 Interactive Effects of Soil Enzymes on OM Accretion

The role of N in SOM decomposition has been studied extensively, leading to the general concept that decomposition is driven by the stoichiometry of the organic compounds and by microbial demand for resources (Swift et al. 1979). This conceptual model predicts that decomposition of an organic compound with a large C:N ratio is N-limited; increasing N availability should enhance decomposition. It appears to apply fairly well to the early stages of decomposition, but poorly to the later stages (Henriksen and Breland 1999; Wang et al. 2004). The distinction between early and late stages of decomposition was attributed to fundamental differences in N availability between labile and recalcitrant components. In contrast to the 'stoichiometry decomposition model', it was hypothesized that N-limited microbes degrade recalcitrant compounds to acquire N, i.e., a "microbial N mining hypothesis" (Moorhead and Sinsabaugh 2006; Craine et al. 2007). Thus, increased N availability should cause an increase in decomposition of labile compounds, but a reduction in the decomposition of recalcitrant compounds, possibly due to the inhibition of oxidative enzymes by N (Kirk and Farrell 1987). An extensive survey of different plant materials by Craine et al. (2007) supports this model. By fitting the decomposition with a two-pool, zero- and first-order mixed model of leaf biomass from >100 plants (grass, forbs, and trees), they demonstrated that mineral N addition stimulated decomposition of labile compounds but inhibited the assault on recalcitrant materials. However, there appeared to be a threshold C:N ratio (i.e., < 15), below which mineral N input had little impact on the decomposition of recalcitrant components of plant materials (Craine et al. 2007). The data from present study show that there was little effect of mineral N input on the decomposition of grass clippings having a C:N ratio of 10.8 supports this (Yao et al. 2009).

11.7 Conclusions

A growing amount of evidence indicates that turfgrasses may be an important part of C sequestration efforts in the U.S. Based on chronosequences from different geographic regions, it seems that soil C accumulation beneath turfgrasses is non-linear, with decreasing net accumulation rates occurring over time. Minimal accumulation rates are approached 25–40 years after turfgrass systems are established. The decreasing net accumulation rates evidently result from increased C degradation by soil microbes. The microbial population in turfgrass soils steadily increases over time, providing the biological infrastructure for degradation of C being generated by turfgrass growth. Sustained C accumulation at low rates in older systems appears to reflect recalcitrant C forms, and C generated by root growth deeper in the soil horizon where microbial activity is lower. Recent studies show that a diverse microbial community is present under turfgrasses, and that community composition is relatively stable as the systems age. The microbial population is associated with a suite of soil enzymes that catalyze degradation reactions; both hydrolytic and oxidative enzyme activities are involved.

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Chapter 12
Using Soil Health Indicators to Follow Carbon Dynamics in Disturbed Urban Environments – A Case Study of Gas Pipeline Right-of-Way Construction

R.R. Schindelbeck and H.M. van Es

Abstract Intense use of urban soils causes perturbations in soil ecosystem functioning. Construction activities often involve soil removal, compaction by heavy equipment, mixing of topsoil and subsoil materials, etc. Urban soil environments share many of the same properties of soils found in other areas of managed and natural systems. The Cornell Soil Health Test (CSHT) can be used as a tool to evaluate these negative impacts through a holistic soil assessment framework which provides an evaluation of indicators of soil physical and biological and chemical processes. The suite of soil tests in the CSHT measures key soil processes. Land managers use the CSHT to first identify the soil quality functional constraints and then adapt soil management to address identified limitations.

The general approach to move from measured soil parameter information through understanding soil functionality to developing a management plan is analogous in farming applications and urban development. By nature this is not a prescriptive approach but instead encourages farmers, land use planners and developers to use this framework to discuss farmland management alternatives, construction scenarios or land reclamation strategies.

A case study is presented to evaluate right-of-way (ROW) construction impacts on soil from the installation of a natural gas delivery pipeline across several land use types- Agricultural (called here Restoration Intensity High (RI-Hi)), Wetland (RI-Wet) and Fallow (Restoration Intensity Low (RI-Lo)). These different land use types have different mandated soil construction and restoration practices. By collecting paired soil health samples directly on the ROW corridor and at adjacent sites just off the ROW we can assess the efficacy of applied soil construction practices and remediation techniques on soil function for each land use type.

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Comparison between ON- and OFF- the right-of-way across land use types showed that wet aggregate stability, total organic carbon content and active carbon were most negatively affected by construction and remediation efforts. The enhanced soil preservation efforts used on the RI-Hi and RI-Wet areas helped maintain soil function. The measured reduction in soil quality parameters, mostly the result of less stringent construction and soil restoration requirements, suggests that the RI-Lo areas would benefit from further reclamation efforts.

Direct understanding of local soil effects can be obtained from the quantitative comparison of each measured soil parameter from the baseline Soil Health Reports using paired samples collected within and outside the ROW corridor. Negatively impacted soil functions are highlighted to guide the application of appropriate soil management and reclamation activities. Repeated sampling in time can be used in a similar manner to measure effectiveness of applied remedial practices.

Soils in the United States are being converted to urban uses at increasing rates – from 0.6 million ha per year in the period 1982–1992 to 0.9 million ha per year during 1992–1997 (Scheyer and Hipple 2005). Vast areas of natural soils are often excavated or disturbed during construction of buildings, neighborhoods and roadways. Soils are dramatically altered by human activities in urban environments and those alterations compromise soil functioning compared to soils in natural or agricultural environments, or within other urban environments (Craul 1999). Construction activities often involve soil removal, compaction by heavy equipment, mixing of topsoil and subsoil materials, etc. Until recently, there were limited tools available to assess these negative impacts through a holistic soil assessment framework, as presented here.

From a landscape perspective, the complex ecological soil system provides for a variety of ecosystem services including water infiltration, storage and filtering, nutrient and carbon storage and cycling, climate regulation, temperature moderation, and organism habitat, among many others. Urban soil environments share many of the same properties of soils found in other areas of managed and natural systems. Human management impairs soil function and exacerbates environmental decline. Roofs, streets and impervious pavements redirect precipitation away from the soil's filtration and storage capabilities, thereby placing greater demands on the remaining soil areas. Excessive disturbance and soil compaction can alter soil structure with reduced infiltration of water leading to increased potential for runoff and erosion. Soil organisms and their related nutrient cycles are disrupted by soil disturbance and chemical inputs.

The concept of soil quality can be viewed as an integration of the physical, biological and chemical components and processes and their interaction (Karlen et al. 2001). Soil physical structure plays a direct role in biological and chemical transformations (Dexter 2004) and is in turn affected by microorganism growth and exudates and organic matter decomposition (Amezketa 1999; Magdoff and van Es 2010). Overall soil performance can be limited by disruption of one or more features of the soil ecological complex.

Soil quality can be assessed by how well a soil performs its varied functions. It cannot be measured directly but we can evaluate measurable properties as indicators of soil processes in support of such functions. In the standard agricultural soil test, a limited number of chemical indicators of plant nutrition are evaluated and recommendations for additions to correct deficiencies are provided. A more holistic approach to soil testing would also provide evaluation of indicators of soil physical and biological processes. The recent decades have shown research interest in providing demonstrable, intuitive soil physical and biological assessments in the field (Doran and Jones 1996; Karlen et al. 1997; USDA 1998) and laboratory (Andrews and Carroll 2001; Gugino et al. 2009).

Keywords Soil function • Cornell soil health test • Adaptive management

12.1 The Cornell Soil Health Test

The Cornell Soil Health Test (CSHT) uses a suite of laboratory and field tests to evaluate soil physical, biological and chemical function at reasonable cost (Idowu et al. 2008). Each soil sample is analyzed for four soil physical indicators, four soil biological indicators and seven soil chemical measures. The indicators selected are relevant to critical soil functional processes (Table 12.1; Moebius et al. 2007).

As part of the sampling protocol for the CSHT, each soil health sample consists of disturbed soil composited from two locations nested within five sites on each land unit. Each subsample is collected from the 0–0.15 m depth, and ideally at field capacity moisture content. At each sampling site within the field, two penetrometer measurements are taken as the only in-field assessment in the test. Collected samples are cooled to 4C to reduce biological activity and facilitate a consistent thermal environment for biological response when the sample is brought to ambient temperatures in the laboratory.

Details of analytical procedures used in the CSHT are found in the Cornell Soil Assessment Training Manual (Gugino et al. 2009). Using a database of 1,500 soil samples, scoring functions were developed for all soil indicators (except texture) to rate test results (Andrews et al. 2004). Different scoring functions were developed for the three main textural classes, sand, silt, and clay, with the soil texture being determined using the rapid and inexpensive method of Kettler et al. (2001). The scoring functions enable a specific indicator value to be converted to a rating from 0 to 100 with 100 being best and 0 being poorest. Thresholds were used to rank the ratings with 0–30 corresponding to a deficiency or constraint, 30–70 corresponding to an intermediate region and 70–100 indicating that the measured value is at an optimal level. This approach was evaluated relative to literature reports for vegetable and agronomic crops and in some cases minor modifications were made. Scoring curves for all indicators are presented in Gugino et al. (2009). Reports for the CSHT use the red/yellow/green color scheme to visually rank the low, medium and high scores from each test to highlight functional constraints.

Table 12.1 Brief descriptions of the Cornell soil health test indicators

Physical

Aggregate Stability: is a measure of the extent to which soil aggregates resist falling apart when wetted and hit by rain drops. Measured using a rain simulation sprinkler that steadily rains on a soil sieve containing a known weight of soil aggregates between 0.5 and 2 mm. Unstable aggregates slake (fall apart) and pass through the sieve. The fraction of soil that remains on the sieve determines the percent aggregate stability.

Available Water Capacity: reflects the quantity of water that a disturbed sample of soil can store for plant use. It is the difference between the water stored at field capacity and the wilting point, and is measured using pressure chambers.

Surface Hardness: is calculated from the maximum soil surface (0–0.15 m depth) penetration resistance (psi) determined using a field compaction tester.

Subsurface Hardness: is a measure of the maximum resistance (in psi) encountered in the soil at the 0.15–0.45 m depth using a field compaction tester.

Biological

Organic Matter: is any material that is derived from living organisms, including plants and soil fauna. The percent OM is determined by loss on ignition, based on the change in weight after a dry soil is exposed to 450C in a furnace.

Active Carbon: is a measure of the fraction of soil organic matter that is readily available as a carbon and energy source for the soil microbial community. The soil sample is mixed with potassium permanganate (deep purple in color) and as it oxidizes the active carbon, the reduced permanganate changes color and is measured using a colorimeter.

Potentially Mineralizable Nitrogen: is the amount of nitrogen that is converted (mineralized) from an organic form to a plant-available inorganic form by the soil microbial community over 7 days in an incubator. It is a measure of the soil biological activity.

Root Health Rating: is a measure of the quality and function of the roots as indicated by color, size, texture and absence of symptoms and damage by root pathogens. Bean seeds are grown in a portion of the soil sample in the greenhouse for 4 weeks. Low ratings (1–3) suggest healthy roots because pathogens are not present at damaging levels and/or are being suppressed by the beneficial microorganisms in the soil.

Chemical

Soil Chemical Composition: a standard soil test analysis package measures levels of pH, plant nutrients and toxic elements. Measured levels are interpreted in the framework of sufficiency and excess but are not crop specific.

12.2 Using Constrained Soil Process Information to Guide Management

Land managers use the CSHT to assess soil quality relative to important soil physical, chemical and biological processes and functions as part of an adaptive management framework. The measured information found in the CSHT Report is the basis for broad-sense recommendations of management practices to improve or stimulate constrained soil processes. These management targets can then be considered singly or in combination with other constraints to develop a management plan. Generalized agronomic strategies for field crop and vegetable production are available in the Cornell Soil Assessment Training Manual (Gugino et al. 2009). The context for developing a strategy for management change recognizes that many factors must be considered outside of a prescriptive approach. Factors such as site location, proximity to hazards and populations, manager's attitude and capacities, economics, regulatory environment, weather, labor availability, and available machinery must all be considered in addressing the measured constraints through on-site practices. In urban environments, management options and land use opportunities can be quite different from agricultural situations. However, the general approach to move from measured soil parameter information through understanding soil functionality to developing a management plan is universal. Planners and developers can use this framework to discuss construction scenarios or reclamation strategies.

Assessing the effects of land use practices and soil disturbance in urban environments is often complicated by the lack of a comparison site that represents a "background" or "normal" scenario. In the case of land affected by construction, this can be addressed by collecting soil samples from before vs. after alteration. If this is not feasible, samples can be collected at the same time from on-site and adjacent off-site locations. Adjacent, unaffected areas often allow for an optimum contrast of "background" soil conditions versus applied practices and soil restoration practices can be evaluated at the time of sampling. Repeated sampling in time can be used to measure the longer-term effectiveness of applied soil remedial practices.

12.3 CASE STUDY – Evaluation of Soil Restoration on Pipeline Right-of-Way Construction

The Cornell Soil Health Test (CSHT) provides a standard and holistic framework for soil quality assessment and was used to evaluate right-of-way (ROW) construction impacts using a test case of the Cornell Combined Heat and Power Project (CCHPP). This involved the installation of a 0.2 m diameter gas delivery line over a 4.8 km length from an interstate transmission line to the Cornell University campus during 2008–2009.

Construction of this right-of way could result in damage to soil quality along its course from construction activities like soil removal, compaction by heavy equipment, mixing of topsoil and subsoil materials, etc., thereby affecting the ability of the disturbed soils to sustain soil functions like plant growth, water infiltration and retention, supporting soil life, etc. Figure 12.1 illustrates the magnitude of the soil disturbance that is typically experienced across the construction landscape. Current guidelines for construction and restoration measures, as well as for post-construction monitoring and remediation, are broad-sense and relatively simplistic, and have limited relation to actual soil functioning. Projects of this size require permitting by the New York State Department of Environmental Conservation, which ensures that sensitive land use types are properly managed during and after construction.

Fig. 12.1 Heavy machinery and soil disturbance during pipeline project right of way construction. *Photo a* Pipe installation in trench bottom across fallow land use type. Note vehicle traffic directly on top of subsoil spoil. *Photo b* Stripped topsoil segregated upslope from pipe in agricultural land use area. Vehicles shown backfilling subsoil spoil before topsoil

Table 12.2 Applied pipeline right of way construction and restoration practices by land use type

	Land use type		
Soil management practice	Wetland	Agricultural land	Fallow land
Install pipe. Replace subsoil from trench onto pipe. Rough grade and remove large stones. Disk harrow.	**X**	**X**	**X**
Install rubber or wooden mats to avoid compaction. Segregate topsoil and trench spoil on top of mats. Remove mats in reverse order.	**X**		
Strip and stockpile topsoil. Deep rip after rough grading of subsoil. Replace topsoil. Rough grade and deep rip again.		**X**	
Apply lime and fertilizer.		**X**	**X**
Broadcast grass seed and apply straw mulch.	**X**		**X**
Drill grass seed and apply straw mulch.		**X**	

Soil quality sampling to monitor and assess the impact of construction activities can ensure that applied construction standards are adequate and suggest a focus for remediation efforts. The CCHPP crosses agricultural land, wetlands and fallow land and the associated generalized New York State construction standards and practices for each land use type are outlined in Table 12.2. The construction and reclamation requirements are most exacting for Agricultural land, called here Restoration Intensity High (RI-Hi), followed by Wetlands (RI-Wet) and finally Restoration Intensity Low (RI-Lo) for Fallow land. A map showing the locations where samples were collected from the different land use areas, in a paired sampling scheme (ON- and OFF-right of way), is shown in Fig. 12.2.

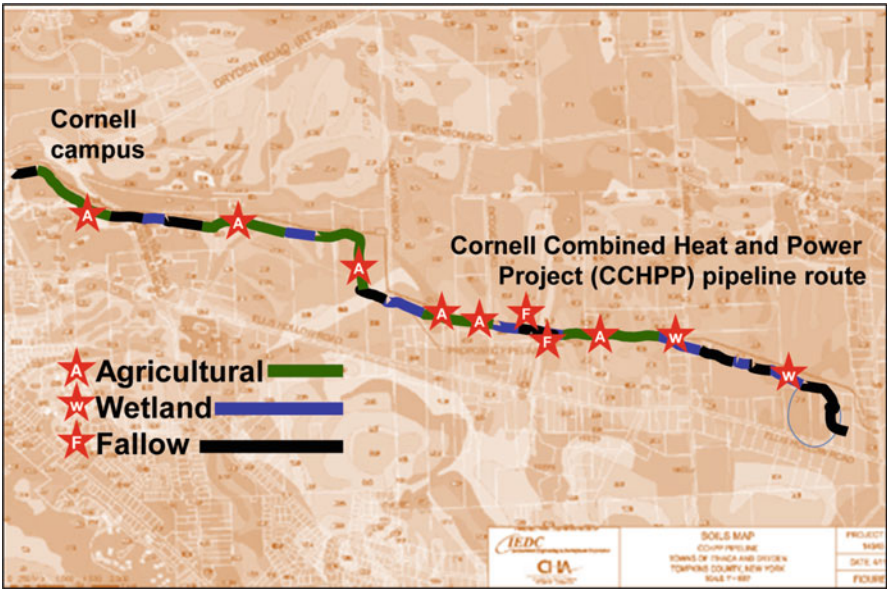

Fig. 12.2 Soils map with land use type and soil health assessment sampling sites (June 2009)

12.3.1 High Intensity Restoration (RI-Hi) Lands

In RI-Hi areas, stockpiled topsoil from the ROW was left in long piles at the edge of the ROW to be spread in the Spring of 2009 (Fig. 12.3, Photo A). This left the rough-graded subsoil exposed to the beneficial effects of winter freeze and thaw cycles. Soil conservation structures including soil surface grading, silt curtains and straw bales were employed in erosion sensitive areas. Dry April 2009 conditions allowed for 0.45 m deep subsoil ripping using an Unverferth (Kalida, OH) zone builder implement (Fig. 12.3, Photo B), followed by large stone removal. This decompaction effort was aimed at loosening the subsoil from the pipe installation and associated heavy vehicle traffic. The stockpiled topsoil was then spread over the ROW and rough graded (Fig. 12.1, Photo B). The area was again sub-soiled before disking, raking and final large stone removal. This was followed by broadcasting moderate amounts of lime (2,268 kg ha^{-1}) and fertilizer (678 kg ha^{-1} 5-10-5). The land was then seed-drilled with a conservation grass seed mix and mulched with straw (Fig. 12.3, Photo C, just before mulching) in June 2009.

12.3.2 Wetland (RI-Wet) Areas

The Wetlands traversed by the ROW represent sensitive natural areas. Pipeline construction practices must minimize adverse effects when crossing these areas as

Fig. 12.3 Representative photos after pipe installation from pipeline project right of way. *Photo a* Agricultural land after pipe burial. Fall, 2008. Topsoil stockpiled on *right*. *Photo b* Unverferth zone-till ripper used to decompact ROW in agricultural land areas. *Photo c* Agricultural land (see *Photo a*) after to drilling of grass mix. Spring, 2009. *Photo d* Wooden timber mats across ROW protect wetland soils. Soil berms and silt curtains control erosion

specified in the construction permit. In the RI-Wet areas, wooden mats were installed ahead of all construction equipment and driven on through all stages of construction (Fig. 12.3, Photo D). Working from the flotation area, the topsoil and trench spoil were segregated while the trench was open. Once the concrete-coated pipe was installed, the trench was backfilled with subsoil. The topsoil was replaced and rough graded. Wooden mats were then removed in reverse order of installation. The ROW in these RI-Wet areas was effectively completed in the Fall of 2008. Grass seed and straw mulch was then broadcast. Lime and fertilizer was spread in the Spring of 2009 and more grass seed was broadcast to facilitate complete ground coverage.

12.3.3 Low Intensity Restoration (RI-Lo) Areas

Lands traversed by the ROW that are not defined as Agricultural or Wetland only had to meet minimum requirements for construction management as defined in the State permit (Table 12.2). In these locations, topsoil was kept segregated from subsoil as trench construction progressed. The subsoil material was often trafficked during pipe installation (Fig. 12.1, Photo A). This subsoil spoil was used to fill the trench. The topsoil was then replaced over the entire ROW width before the topsoil was rough graded. These areas were then seeded and mulched in the Fall of 2008. Lime and fertilizer was spread in the Spring of 2009 with more grass seed broadcast and mulch spread where necessary.

12.4 Post-construction ROW Soil Health Assessment 2009

12.4.1 Soil Sample Collection

Soil samples were collected from the pipeline construction area in June, 2009. The conservation grass mix had established well and was holding the soil against erosion. Sampling sites were chosen to capture typical characteristics of each soil mapping unit encountered – RI-Hi, RI-Wet and RI-Lo. Composite soil samples were collected from locations directly on the ROW and just off the ROW in the adjacent undisturbed land area. Each location therefore has paired samples from within the disturbed area and an associated "benchmark" sample of unaffected soil collected from directly outside the construction area. Comparison of collected data from the paired locations allowed for immediate quantitative evaluation of the effects of construction on soil function and the efficacy of the varying construction practices and remediation techniques applied to the different land use types. The representative sampling of soil from the 0 to 0.15 m depth is coupled to soil hardness readings at 0 to 0.15 m and 0.15 to 0.45 m depth using a Dickey-John (manufacturer) soil penetrometer.

12.4.2 Analysis of 2009 Soil Health Indicator Data

Samples from ten sites with paired samples collected on- and off- the ROW were submitted to the Cornell Soil Health Testing Laboratory for analyses according to the CSHT protocol (Table 12.1). Mean measured values for each of the land use types for ON vs. OFF ROW locations are presented in Table 12.3, as well as the P-value for a test of significant change. The most significant effect was measured for aggregate stability, a measure of soil structural resiliency, which showed a significant reduction for all land use types. Combining all land types showed a 32% reduction due to construction activities. This indicator is sensitive to soil perturbations which lead to surface crusting, reduced water infiltration and hardness. The available water capacity values remained relatively similar after construction, suggesting that the construction efforts, including topsoil stockpiling separate from clay-rich subsoil spoil

Table 12.3 Mean values from On and Off the right of way, difference between values and significance level by land use type, and all data combined

	Agricultural land (n = 12)				Wetland (n = 4)				Fallow land (n = 4)				Combined (n = 20)			
	ON	OFF	Diff	(p value)	ON	OFF	Diff	(p value)	ON	OFF	Diff	(p value)	ON	OFF	Diff	(p value)
Aggregate stability (%)	57	78	−21	0.003	74	90	−16	0.025	36	89	−53	0.060	56	83	−27	0.000
Available water capacity (m/m)	0.15	0.16	−0.01	p> 0.1	0.21	0.21	0.00	p> 0.1	0.16	0.18	−0.02	p> 0.1	0.17	0.17	0.00	0.090
Surface hardness (MPa)	0.92	0.95	−0.03	p> 0.1	0.88	0.74	0.14	p> 0.1	2.49	1.73	0.76	0.004	1.23	1.06	0.17	p> 0.1
Subsurface hardness (MPa)	2.16	2.44	−0.28	0.108	1.86	2.17	−0.31	p> 0.1	2.93	2.61	0.32	p> 0.1	2.26	2.42	−0.16	p> 0.1
Organic carbon g kg^{-1}	27	32	−5	0.079	25	26	−1	p> 0.1	15	31	−17	0.043	24	43	−19	0.017
Active C mg kg^{-1}	675	722	−47	p> 0.1	602	656	−54	0.067	414	606	−193	0.022	608	685	−77	0.060
Potentially mineralizable nitrogen (μgN/gdwsoil/week)	17.8	17.7	0.1	p> 0.1	17.5	18.6	−1.1	p> 0.1	8.4	31.6	−23.2	0.090	15.9	20.7	−4.8	0.099
Root health rating (1–9)	4.2	4.1	0.1	p> 0.1	3.3	3.6	−0.3	p> 0.1	3.1	3.6	−0.5	p> 0.1	3.8	3.9	−0.1	p> 0.1
pH	6.2	6.0	0.2	p> 0.1	6.1	5.8	0.3	p> 0.1	7.2	5.1	2.1	0.015	6.3	5.8	0.6	0.039
Extractable phosphorus (ppm)	15.7	12.8	2.9	p> 0.1	2.0	2.0	0.0	p> 0.1	1.5	1.5	0.0	p> 0.1	10.1	8.4	1.7	p> 0.1
Extractable potassium (ppm)	162.5	132.5	30.0	0.052	95.0	90.0	5.0	p> 0.1	65.0	135.0	−70.0	0.088	129.5	124.5	5.0	p> 0.1
Magnesium (ppm)	356.7	331.7	25.0	p> 0.1	330.0	413.0	−83.0	p> 0.1	225.0	92.5	132.5	0.042	325.0	300.0	25.0	p> 0.1
Iron (ppm)	20.8	38.0	−17.2	0.088	26.0	21.0	5.0	p> 0.1	22.5	44.0	−21.5	p> 0.1	22.2	35.7	−13.5	0.053
Manganese (ppm)	23.3	22.3	1.0	p> 0.1	42.0	45.5	−3.5	p> 0.1	46.5	13.0	33.5	0.079	31.7	25.1	6.6	p> 0.1
Zinc (ppm)	0.7	1.1	−0.4	0.036	0.9	1.7	−0.8	p> 0.1	0.3	1.4	−1.1	p> 0.1	0.6	1.2	−0.6	0.002
overall SH score	73.5	73.5	0.0	p> 0.1	73.6	70.3	3.3	p> 0.1	42.6	58.5	−15.9	0.070	67.3	69.9	−2.6	p> 0.1

Table 12.4 Percent change in indicator values of selected soil health indicators, as compared to benchmark locations OFF the pipeline right of way

	Agricultural land (n = 12)				Fallow land (n = 4)			
	ON	OFF	Diff	Percent change	ON	OFF	Diff	Percent change
Aggregate stability (%)	57	78	−21	−27	36	89	−53	−60
Available water capacity (m/m)	0.15	0.16	−0.01	ns	0.16	0.18	−0.02	ns
Surface hardness (MPa)	0.92	0.95	−0.03	−3	2.49	1.73	0.76	+44
Subsurface hardness (MPa)	2.16	2.44	−0.28	−11	2.93	2.61	0.32	+12
Organic carbon g kg^{-1}	27	32	−5	−16	15	31	−17	−55
Active C mg kg^{-1}	675	722	−47	−7	414	606	−193	−32
Potentially mineralizable nitrogen (μgN/gdwsoil/week)	17.8	17.7	0.1	ns	8.4	31.6	−23.2	−73
Overall SH score	73.5	73.5	0.0	ns	42.6	58.5	−15.9	−27

Agricultural and fallow land use types

had minimal impacts on this indicator. The use of wooden mats during construction over Wetlands effectively controlled soil surface and subsurface compaction. The reclamation practice of soil decompaction after construction in RI-Hi areas limited surface compaction and controlled subsurface compaction culminating in an overall reduction in surface and subsoil hardness compared to the native condition. The lack of enhanced surface soil handling restrictions or reclamation practices in the RI-Lo land areas resulted in increased surface soil hardness. Subsurface soil hardness was also increased in the RI-Lo areas but not significantly.

Significant negative impacts were measured for total organic carbon even where topsoil was separated and stockpiled during construction in the RI-Hi land areas. Overall, total organic carbon levels were decreased by 44% compared to the benchmark locations. The organic carbon levels in the RI-Wet sites benefitted from the controlled trafficking patterns on the soil. Where minimal soil protection and reclamation was performed on the RI-Lo sites, total soil carbon was decreased by 55% (Table 12.4). Biologically active carbon on the right of way after completion of the project was significantly reduced across all sites compared to locations off the right of way. The potentially stimulating effect of soil stirring and mixing on soil biological activity was mitigated by the negative effects of the heavy machinery. The rate of soil biological activity measured by the PMN test was decreased by 73% in the RI-Lo areas (Table 12.4) possibly due to uncontrolled mixing of biologically active topsoil with nutrient-poor subsoil.

The application of moderate rates of lime and agricultural fertilizer increased the levels of the applied nutrients where it was applied ON the right of way. The increased pH from the applied lime reduced the levels of the metal micro-nutrients Fe and Zn on the RI-Hi land. The RI-Wet sites did not receive any chemical amendments.

12.4.3 Benchmark Comparison

Direct understanding of local pipeline construction effects can be seen in the comparison of the baseline reports from the paired samples from within and outside the ROW corridor. Each site has two sampling locations associated with it- one directly on the ROW and the other just off the ROW in undisturbed soil representing the ROW before construction. A comparison of paired samples allows for immediate quantitative evaluation of the effects of construction on native soil function. The paired Cornell Soil Health Reports shown in Fig. 12.4 use a red, yellow and green color scheme to rank the indicator values into categories of low, medium and high. The red color reveals a soil functional constraint that can be targeted by management. The right of way construction shows the aggregate stability, soil organic matter and PMN values to decrease from the optimum range to a deficient-constrained range.

An example of an RI-Lo location showing marked negative effects of the pipeline construction efforts is presented in Fig. 12.4. Comparing these paired samples shows that the construction activities reduced aggregate stability by 63%, available water capacity by 2% and increased the surface and subsurface hardness by 43% and 17% respectively. Organic matter levels were decreased 50%, biologically active carbon was reduced 23% and the soil potentially mineralizable nitrogen capacity was reduced by 76%. The application of lime and fertilizer on the right of way after surface smoothing increased soil pH and phosphorus levels over the native soil.

The percent change from the adjacent undisturbed soil for selected soil health indicator values from the RI-Hi and RI-Lo locations is shown in Table 12.4. The soil parameter effects from enhanced surface soil protection during pipeline construction and additional restoration efforts in the RI-Hi locations can be compared to the minimal efforts in soil management in the RI-Lo areas. Table 12.4 shows that aggregate stability decreased half as much in the RI-Hi areas compared to RI-Lo (27% versus 60%). Protection of topsoil and subsequent deep ripping after soil replacement actually decreased soil hardness in the surface and subsurface of the RI-Hi areas (−3% and −11% respectively). The RI-Lo land saw a 44% and 12% increase in surface and subsurface soil hardness, respectively.

Multiple soil biological indicators were seen to decrease when comparing samples collected on and collected outside the construction zone (Table 12.4). At the RI-Hi sites, construction activities decreased soil organic carbon by 16% and active carbon by 7%. The RI-Lo sites showed a 55% decrease in organic carbon, 32% reduction in the level of active carbon and a 73% reduction in biological activity as measured in the PMN analysis. The overall soil quality score was unaffected by the heavy construction and associated restoration practices applied in the RI-Hi sites. The limited post-construction remedial activities in the RI-Lo zones did not alleviate negative effects on soil function in these areas. The overall soil quality scores in the RI-Lo sites decreased significantly, averaging 27%.

CORNELL SOIL HEALTH TEST REPORT — Sample ID: G883

Name of Farmer: Cornell Univ Utilities	Sample ID: G883
Location: CCHPP Pipeline Project	Agent: Bob Schindelbeck, Dept. of Crop and Soil Science, CU
Field/Treatment: BERKLEY ON RoW	Agent's Eimal: 0
Tillage: FALLOW Land Use Type-Minimal Soil Restoration Prac	Given Soil Texture: 0
Crops Grown: Grass cover	Date Sampled: 6/14/2009

	Indicators	Value	Rating	Constraint
PHYSICAL	Aggregate Stability (%)	27.5	35	
	Available Water Capacity (m/m)	0.13	28	water retention
	Surface Hardness (psi)	375	1	rooting, water transmission
	Subsurface Hardness (psi)	450	5	Subsurface Pan/Deep Compaction
BIOLOGICAL	Organic Matter (%)	2.5	22	energy storage, C sequestration, water retention
	Active Carbon (ppm) [Permanganate Oxidizable]	509	29	Soil Biological Activity
	Potentially Mineralizable Nitrogen (µgN/ gdwsoil/week)	5.2	0	N Supply Capacity
	Root Health Rating (1-9)	3.0	75	
CHEMICAL	pH (see Nutrient Analysis Report)	7.5	67	
	Extractable Phosphorus (see Nutrient Analysis Report)	2.0	44	
	Extractable Potassium (see Nutrient Analysis Report)	65.0	100	
	Minor Elements (see Nutrient Analysis Report)		56	
	OVERALL QUALITY SCORE (OUT OF 100):		38.5	Very Low

Soil Textural Class:=> silt loam
SAND (%):37.3 SILT (%): 54.3 CLAY (%): 8.4

CORNELL SOIL HEALTH TEST REPORT — Sample ID: G884

Name of Farmer: Cornell Univ Utilities	Sample ID: G884
Location: CCHPP Pipeline Project	Agent: Bob Schindelbeck, Dept. of Crop and Soil Science, CU
Field/Treatment: BERKLEY OFF RoW	Agent's Email: 0
Tillage: FALLOW Land Use Type-Minimal Soil Restoration Prac	Given Soil Texture: 0
Crops Grown: Grass cover	Date Sampled: 6/14/2009

	Indicators	Value	Rating	Constraint
PHYSICAL	Aggregate Stability (%)	90.4	100	
	Available Water Capacity (m/m)	0.16	49	
	Surface Hardness (psi)	263	15	rooting, water transmission
	Subsurface Hardness (psi)	383	17	Subsurface Pan/Deep Compaction
BIOLOGICAL	Organic Matter (%)	5.0	87	
	Active Carbon (ppm) [Permanganate Oxidizable]	689	66	
	Potentially Mineralizable Nitrogen (µgN/ gdwsoil/week)	21.6	100	
	Root Health Rating (1-9)	2.8	88	
CHEMICAL	pH (see Nutrient Analysis Report)	5.3	0	Toxicity, Nutrient Availability (for crop specific guide, see CNAL report)
	Extractable Phosphorus (see Nutrient Analysis Report)	1.0	17	<4.5:Plant P Availability, >25: Env. Loss Potential
	Extractable Potassium (see Nutrient Analysis Report)	115.0	100	
	Minor Elements (see Nutrient Analysis Report)		56	
	OVERALL QUALITY SCORE (OUT OF 100):		57.9	Medium

Soil Textural Class:=> silt loam
SAND (%):24.2 SILT (%): 68.4 CLAY (%): 7.4

Fig. 12.4 Cornell soil health test reports from a paired fallow land use type sample location

12.5 Conclusions

Holistic soil health tests allow for an integrated assessment of the functional capacity of the soil at each location and targeted evaluation of past and future construction and restoration practices. Comparison between ON- and OFF- the right-of-way across land use types showed that wet aggregate stability, total organic carbon content and active carbon were most negatively affected by construction and remediation efforts.

Examining the effects of construction and remediation practices suggested that the enhanced soil preservation efforts used on the RI-Hi and RI-Wet areas helped maintain soil function. The measured reduction in soil quality, mostly the result of less stringent construction and soil restoration requirements, suggests that the RI-Lo land areas would benefit from further reclamation efforts.

Direct understanding of local soil effects can be obtained from the quantitative comparison of each measured soil parameter from the soil health reports from paired samples collected within and outside the ROW corridor. Negatively impacted soil processes are highlighted to guide the application of appropriate soil management and reclamation activities. Repeated sampling in time can be used in a similar manner to measure effectiveness of applied remedial practices.

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Chapter 13
Carbon Sequestration in Golf Course Turfgrass Systems and Recommendations for the Enhancement of Climate Change Mitigation Potential

Adam L. Selhorst and Rattan Lal

Abstract Urban ecosystems are an important and ever growing land use throughout the U.S. and globally. Characterized by large growing areas and intensive management, urban ecosystems play an important role in the global carbon (C) cycle. Thus, soil C budgets were constructed for several golf courses in Central Ohio. The experimental data show that golf turfgrasses sequestered C to a depth of 15 cm. Soils in the top 2.5 cm sequestered C for 14 years in fairways and 12 years in rough sites, increasing to as much as 81 years in fairways and 91.4 years in rough at a depth of 10–15 cm.

The hidden C costs (HCCs) of golf course development were also assessed and C emissions per year were determined. Major C emissions were attributed to nitrogen (N) fertilizer use (1,498 kg Ce(Carbon Equivalent)/year), fungicide application (1,377 kg Ce/year), unleaded fuel burning (3,618 kg Ce/year), diesel fuel burning (6,557 kg Ce/year), and irrigation (626 kg Ce/year). Overall emissions per year for golf course maintenance were estimated at 14.15 Mg Ce/year.

In general, golf courses had a mean C sequestration rate of 3.6 Mg C/ha/year in fairways and 2.5 Mg C/ha/year in rough soils, translating into a total C sequestration potential of 3,517 Mg C for a newly constructed course. When C emissions due to HCCs are subtracted from gross sequestration, the net sequestration is 1610.7 Mg C. Extrapolating data to the entire country, golf turfgrass systems in the U.S. have a technical potential to sequester up to 28.7 Tg C once all current courses have attained the equilibrium level.

Due to the large emission levels created by maintenance practices, however, each course shifts from being a sink to a source after 30.4 years, as emissions by this time exceed the sequestration levels, and continue to remain so for the life of the course. Thus, C sequestered is negated though management emissions 205 years after soil C sequestration had reached the equilibrium level.

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Turfgrass systems have a large technical potential of C sequestration. However, the HCCs of input diminish the net potential. Thus, management practices must be reevaluated to abate the large emissions created and to achieve the benefits of global turfgrass systems in reducing the net anthropogenic emissions.

Keywords Sustainable management • Recreational land use • Urban soils • Climate change • Carbon sequestration • Turf soils • Urbanization

List of Abbreviations

BMPs	Best Management Practices
C	Carbon
Ce	Carbon Equivalent
CO_2	Carbon Dioxide
HCCs	Hidden Carbon Costs
K	Potassium
N	Nitrogen
OCC	Oakhurst Country Club
P	Phosphorous
SOC	Soil Organic Carbon
SOM	Soil Organic Matter

13.1 Introduction

Land use and land use change strongly affect both the global carbon (C) budget, and its ecosystem services. With accelerated global urbanization, land use change can lead to a host of negative environmental impacts including the disruption and depletion of soil organic matter (SOM) pools resulting in the direct release of C into the atmosphere as carbon dioxide (CO_2) (Schlesinger 2000). Especially large fluxes of CO_2 are observed in forest degradation (Brown et al. 1994; Flint and Richards 1991, 1994; Houghton 1991; Houghton and Hackler 1994), releasing between 25% and 30% of their C pools (Detwiler 1986; Schlesinger 1986; Davidson and Ackerman 1993), as well as in wetland destruction which can release much of the estimated 29 g C/m^2/year they sequester (Lal et al. 1995).

Deforestation and destruction of these vital ecosystems is occurring in large part due to the ever increasing global population and subsequent urbanization. Over the last decade the U.S. population alone has increased by almost 27 million (Table 13.1). Due in part to this increase, in the same decade major metropolitan cities have grown by an average of more than 66,000 individuals with a total of more than 24 million Americans moving into urbanized areas (Table 13.2). By the year 2000, urban land in the U.S. was estimated to occupy between 3.5% and 4.9% of the land area or

Table 13.1 Change in population for major regions over a 10 year period for the United States

U.S. Region	Population (July 2000)	Population (July 2010)	Change
Northeast	53,663,333	55,417,311	1,753,978
Midwest	64,491,889	66,972,887	2,480,998
South	100,559,291	114,404,435	13,845,144
West	63,451,331	72,256,183	8,804,852
United States	282,165,844	309,050,816	26,884,972

Adapted from the U.S. Census Bureau (2011a)

Table 13.2 Population trends for the 15 largest metropolitan areas within the United States over a 10 year period

Metropolitan city	Population (July 2010)	Population change (2000–2010)	Change (%)
New York-Northern NJ-Long Island, NY-NJ-PA	19,069,796	746,357	4.1
Los Angeles-Long Beach-Santa Ana, CA	12,874,797	509,169	4.1
Chicago-Naperville-Joliet, IL-IN-WI	9,580,567	481,937	5.3
Dallas-Fort Worth-Arlington, TX	6,447,615	1,286,078	24.9
Philadelphia-Camden-Wilmington, PA-NJ-MD	5,968,252	281,094	4.9
Houston-Sugar Land-Baytown, TX	5,867,489	1,152,072	24.4
Miami-Fort Lauderdale-Pompano Beach, FL	5,547,051	539,059	10.8
Washington-Arlington-Alexandria, DC-VA-MD	5,476,241	680,167	14.2
Atlanta-Sandy Springs-Marietta, GA	5,475,213	1,227,192	28.9
Boston-Cambridge-Quincy, MA-NH	4,588,680	196,331	4.5
Detroit-Warren-Livonia, MI	4,403,437	−49,121	−1.1
Phoenix-Mesa-Scottsdale, AZ	4,364,094	1,112,206	34.2
San Francisco-Oakland-Fremont, CA	4,317,853	194,108	4.7
Riverside-San Bernardino-Ontario, CA	4,143,113	888,296	27.3
Seattle-Tacoma-Bellevue, WA	3,407,848	363,951	12.0
Average All Metropolitan U.S. Cities	**703,156**	**66,345**	**10.0**
Total All Metropolitan U.S. Cities	**257,355,190**	**24,282,357**	**10.4**

Adapted from U.S. Census Bureau (2011b)

roughly 40.6 Mha (Nowak et al. 2001; Robbins and Birkenholtz 2003). The 15 year period between 1982 and 1997 saw an increase in land devoted to urban development of 34%, most directly resulting in accelerated deforestation. Accounting for continued population growth and economic affluence, it is estimated that by 2025 U.S. urbanized land area will increase by 79% raising the urban land area to as much as 9.2% of the total land area of the U.S. (Alig et al. 2004).

Turfgrass soils are a dominant vegetation type and land use in urban ecosystems, which in many regions require extensive management. Americans alone spend on average $25 billion/year on turf development and maintenance (Cockerham and Gibeault 1985). Due to this fact, turfgrass systems are often associated with detrimental environmental impacts. One such impact is the leaching of applied chemical fertilizers and pesticides. Nitrogenous fertilizer and chemical-based pesticides are applied to enhance turf growth. However, such chemicals are often highly mobile and can be directly leached into groundwater or nearby waterways. Such transport into natural waters can result in both groundwater contamination as well as accelerate further use of fertilizers and pesticides. Leaching and runoff potential are dependent on a variety of soil and climatic conditions but may lead to losses of as much as 53% of the applied N fertilizers and 875 g/ha/year for applied pesticides (Haith and Duffany 2007; Petrovic 1990). The leaching of these chemicals can be further accelerated by what is often considered another externality of turfgrass development, ongoing irrigation. Especially in hot dry climates, turfgrasses have a substantial water requirement. This has often been seen as one of the major negatives of turf development with an 18 hole golf course using an estimated 2.3 million liters of water per day (UNESCO 2003).

Despite some of their highly publicized downsides, turf systems also provide a host of environmental benefits, including soil erosion control, heat dissipation, flood control, improved groundwater recharge potential, and recreational area to name a few (Beard and Green 1994). In addition, they may also be relevant to mitigation of the global climate through their ability to sequester C over time. If just 10% of the 120 Pg C annually photosynthesized gross primary productivity is retained in soils and the biosphere, it would be enough to completely offset anthropogenic emissions (Lal 2004). While it is unlikely that turf soils and land use could offset global emissions, there exists a large potential to sequester 0.7–1.0 Mg C/ha/year for periods of up to 40 years (Jo and McPherson 1995; Qian and Follett 2002; Bandaranayake et al. 2003; Huh et al. 2008; Pouyat et al. 2009). Thus conversion to turf systems may be beneficial in slowing atmospheric C enrichment under certain conditions. As previously noted, deforestation and wetland destruction allow for massive C emissions and lead to the depletion of stable soil organic carbon (SOC) pools. Thus if turf soils are developed on previously undisturbed wetland or forest ecosystems, the emissions caused by the destruction of such ecosystems are no doubt much greater than the sequestration by the development of turf. However, the conversion of natural ecosystems to agricultural land use can greatly deplete SOC pools and create large areas of C-depleted soils (Burke et al. 1989; Davidson and Ackerman 1993; Gebhart et al. 1994; Buyanovsky and Wagner 1998; Lal and Bruce 1999). Therefore, turfs developed on previously depleted agricultural soils may hold a greater potential to sequester C as the antecedent conditions allow turfgrass soils to re-sequester much of the depleted SOC pool (Burke et al. 1995; Lal et al. 2000; Follett et al. 2001a, b; Mensah et al. 2003).

Even with their potential to sequester large amounts of C in hither to disturbed soils, turfgrass maintenance practices are C-intensive processes which may decrease the overall potential of reducing and off-setting anthropogenic emissions. The use of fossil fuel dependent maintenance practices (i.e., as pesticide use, fertilizer application, and mowing) can be equivalent to or even exceed those for corn (*Zea mays L.*) production thus limiting the benefits of turf C sequestration (Falk 1976). However, with proper planning and execution, it may be possible to minimize the development and maintenance emissions and maximize the net potential of turfgrass C sequestration. The goal of reducing emissions can be realized by utilizing best management practices (BMPs) which not only increase the climate change mitigation potential of turf soils but also greatly reduce many of the existing negative externalities (i.e., pesticide and nutrient runoff) (Baird et al. 2000). Thus, the adoption of BMPs may help to eliminate or diminish many of the environmental impacts of turf development while increasing their overall potential to sequester excess atmospheric C.

For these reasons the objectives of the present study were to:

1. Determine the ability of golf turfgrass systems in Central Ohio to sequester C
2. Determine the potential of golf turfgrasses to off-set emissions in lieu of the maintenance emissions or hidden C costs, and
3. Provide management recommendations to increase the overall C sequestration potential of golf turfgrass systems

This study was based on the hypotheses that

1. Adoption of BMPs can make turfgrass systems a net C sink
2. The indiscriminate use of chemicals and diesel-based inputs can negate the C sequestered in soils

13.2 Methods

Soil samples were obtained from the fairway and rough areas of 11 golf courses as well as from 11 paired agricultural sites within 1.6 km radius in Central OH, USA (Fig. 13.1). Sites were located within the till plains region of OH, and were sited under similar climatic conditions. All courses contained defined fairways where grass was mowed to a significantly lower height than the neighboring turf. Fairway area consisted of all turf mowed at this lower height as well as all greens and tees. While recognizing differences in management, greens are considered similar to fairways in this study. Sampling of greens was not possible, cover less than 1% of all fairway area. Further, their composition and maintenance more closely resembles fairway than rough turf. Rough comprise of turf area located within the boundaries of the course that is not a part of a fairway, green, or tee box. Figure 13.2 shows an aerial view of one golf course hole with fairway area outlined. Oakhurst Country Club (OCC) was used for full emission analysis. It is a private country club located at 39° 53′5.85″ N latitude and 83° 9′27.18″ W longitude, in Grove City, OH, U.S.A.

Two 5.4 cm soil cores were obtained to 15 cm depth from 3 randomly selected fairway and rough soils for each of the 11 courses and 11 corn-based agricultural sites between June 14 and July 8, 2006. Courses used had age chronosequences ranging from 1 to 97 years since conversion from cropland to turfgrass. Each soil core was divided into four depths; 0–2.5 cm, 2.5–5.0 cm, 5.0–10.0 cm, and 10.0–15.0 cm, air dried, ground, sieved through 2 mm and, ball milled prior to determination of SOC concentration using the dry combustion method (Nelson and Sommers 1996) with an NC 2100 soil analyzer (ThermoQuest CE Instruments, Milan, Italy). All soils sampled had $pH < 7.3$, so total C is assumed to be SOC (Golubiewski 2006).

Using the SAS Statistical Procedures (SAS Institute Inc. 1994), nonlinear regressions of SOC concentration by time since turfgrass development were computed for each depth. Slopes of each regression were utilized to determine the maximum C

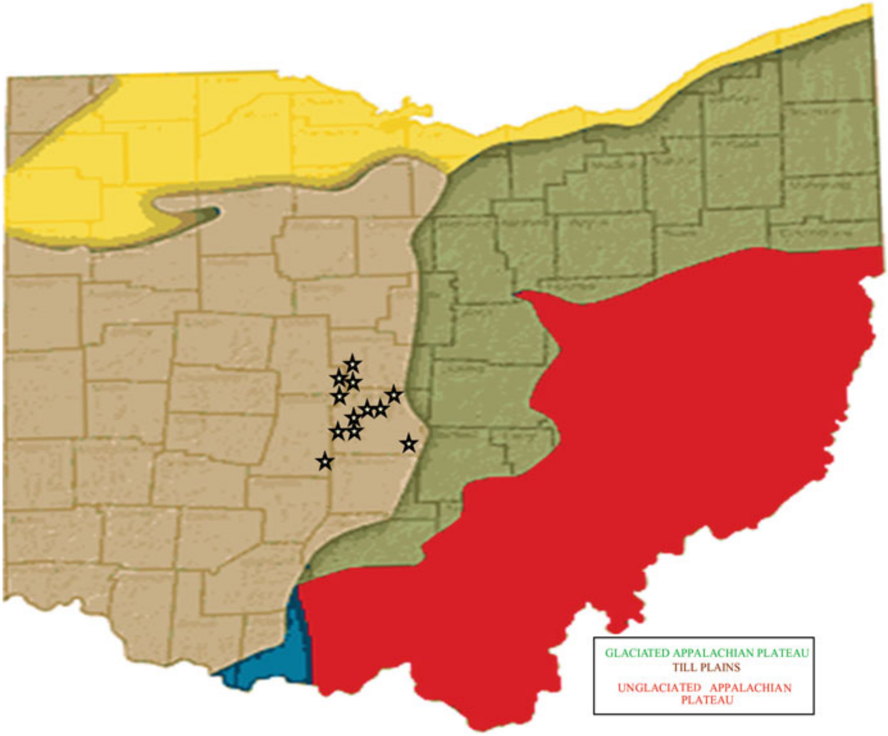

Fig. 13.1 Map of soil sampling sites utilized

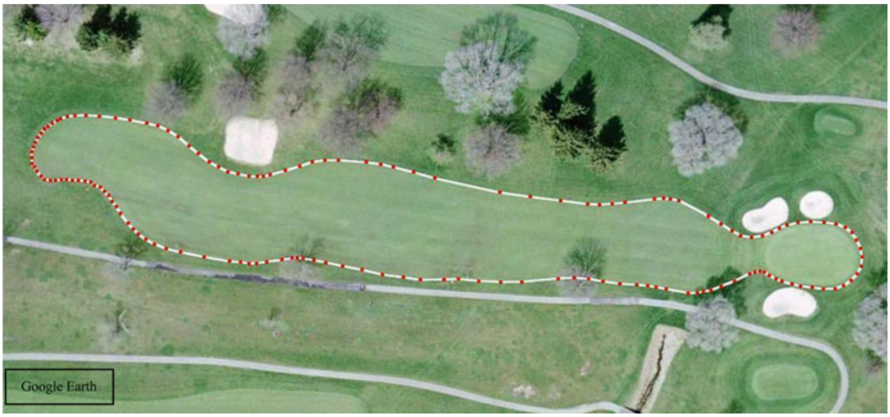

Fig. 13.2 Sample golf course hole with fairway region circled

sequestration rate at each depth, and the final SOC concentration values were utilized to determine total annual sequestration rate, for each depth, and the time needed to reach equilibrium. Minitab 14.1 was used to conduct ANOVAs to determine significant differences between SOC (%) and depth, SOC (%) and years since turfgrass development, SOC (%) and site, and to conduct two-sample t tests to determine significant differences between fairway or rough sites and their corresponding paired agricultural sites.

Five year mean gasoline and diesel fuel use was obtained from long term maintenance data at OCC and converted to kg of C equivalent (Ce) emitted each year utilizing data from Lal (2004). Irrigation information was obtained using the long term maintenance records, and the total irrigation emission were determined (Pira 1997; Whiffen 1991). Amount of N, P and K fertilizer as well as active ingredients from fungicides, insecticides, and pesticides added over a 1 year period were determined from long term maintenance data and converted to kg of Ce (Lal and Bruce 1999). The overall area of rough and fairway for each course was determined using ArcGIS software. The overall C sequestered per site at each depth was then determined using Eqs. 13.1–13.3:

$$10^4 (\mathrm{m}^2 / \mathrm{ha}) * \mathrm{depth(m)} * \rho_b, (\mathrm{Mg/m}^3) * \mathrm{C\ seq/year(\%)/100} = \mathrm{Mg\ C/ha/year} \quad (13.1)$$

$$\mathrm{Mg\ C/ha} * \mathrm{ha} = \mathrm{Mg\ C\ per\ site} \quad (13.2)$$

$$\mathrm{Mg\ C\ per\ site} * \mathrm{years\ of\ sequestration} = \mathrm{Total\ Mg\ C\ sequestered\ per\ site} \quad (13.3)$$

Sequestration and emission data as well as the average course area were then utilized to determine the potential of courses to mitigate atmospheric C enrichment through SOC sequestration. The SOC sequestration potential is defined as the capacity of a turfgrass soil to sequester more C than is being emitted through the practices necessary to maintain such a system. It is the net C-gain in the turf ecosystem.

13.3 Results

13.3.1 Carbon Sequestration in Soils

Data analysis showed that golf turfgrasses sequester C for as little as 12 years and as long as 91 years, depending on depth and type of soil management. The surface layer sequestered C at higher rates in both the fairway and rough soils but equilibrated much faster than sub-soils. In the top 5 cm, SOC sequestration occurred for approximately 12–30 years following the turf establishment. In the lower 10 cm, however, SOC sequestration occurred for as much as 91.4 years or longer. This pattern is most likely due to the decrease in root biomass at deeper depths, and also lower levels of decomposition and C mineralization in the sub-soil. Although time to attain the equilibrium SOC level is shorter in surface soils, the top 2.5 cm layer attains SOC

Table 13.3 Mean carbon sequestration per hectare per year and average sequestration per course per year per course for both fairway, rough, and entire courses for all 18 hole courses utilized in this study

Site	Seq/ha/year (Mg of C)	Seq/site/year (Mg of C)
Fairway	3.55 ± .08	43.6 ± 6.8
Rough	2.64 ± .06	113.4 ± 13.71
Total site		157.1 ± 18.20
Interaction	$P < 0.001$	

One-way ANOVA with Tukey's post test for all interactions of sequestration per hectare per year by site, $p < 0.001$, $F = 857.33$, $DF = 29$. $N = 10$ for each site

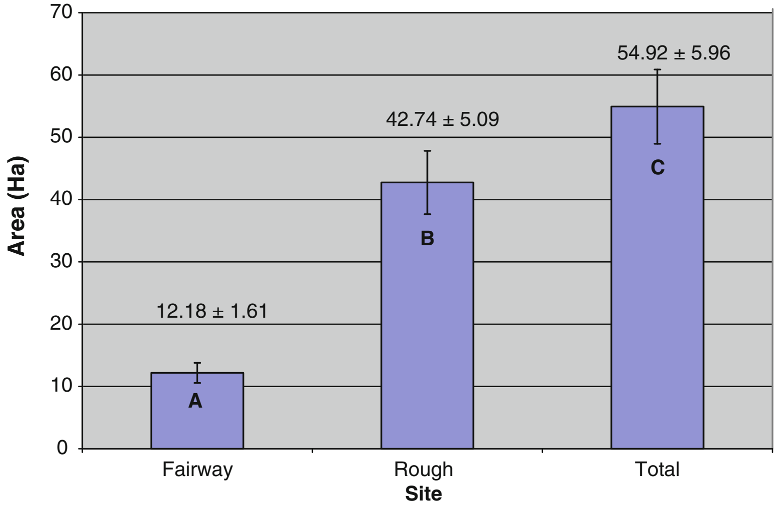

Fig. 13.3 Mean golf course area for fairways, rough, and total course area for all 18 hole courses utilized in this study (One-way ANOVA with Tukey's post test, for all interactions $p < 0.001$, $F = 57.53$, $DF = 29$, means with different letters are significantly different at $p < 0.001$. $N = 10$ for each site. Error bars = 95% CI)

concentrations nearly three times higher than those at 5–15 cm depth in fairway soils and nearly double in rough soils. Estimates of mean SOC sequestration rates range from 3.55 ± 0.08 Mg C/ha/year in fairway soils and 2.64 ± 0.06 Mg C/ha/year in rough soils (Table 13.3). Relatively higher SOC sequestration in fairway than rough soils is most likely due to the increased management that the fairway soils receive. Being the primary playing surface, fairways receive more fertilizers, pesticides, water, and overall care than do rough soils. Although the fairway soils had higher mean sequestration rates, the amount of net C sequestered per site each year was greater in rough soils (113.4 ± 13.71 Mg) than in fairway soils (43.6 ± 6.8 Mg) due to the significantly greater area of rough (42.74 ± 5.09 ha) than fairway managed soils (12.18 ± 1.61 ha) per course (Table 13.3; Fig. 13.3).

13.3.2 Effect of Turfgrass Maintenance and Carbon Emissions

Significant C emissions observed may be attributed to a range of turf maintenance practices (i.e., mowing, fertilizer/pesticide application, and irrigation). Applications of nitrogenous fertilizer had the largest contribution to the HCC (1,498 kg Ce) followed by the potassium (K) fertilizer (138 kg Ce) and finally the phosphorous (P) fertilizer (96.3 kg Ce). Nitrogen fertilizers are by far the most highly used and C-intensive. While P and K fertilizers emit comparable levels of C per kg of fertilizer, K fertilizers are much more widely used, leading to their overall greater contribution.

Use of pesticides also has high HCC. Fungicides emit the largest amount of C per year (1,377 kg Ce) followed by herbicides (206 kg Ce) and insecticides (353 kg Ce). Despite having low HCC per kg of active ingredient the annual rate of application of fungicides is thrice as high as those of herbicides or insecticides. Fungicides also contain much higher levels of active ingredient than do herbicides or insecticides again leading to high HCC.

Direct use of fuel was among the most C intensive management emissions, with diesel being utilized the most (6,975 kg/year) and the most C intensive (6,557 kg Ce). Nevertheless, gasoline use is also high with HCC of approximately 3,618 kg Ce/year.

Finally, irrigation is also a significant source of emissions as courses often use water excessively especially during the hot dry summer months. When factoring in the pumping time, irrigation is estimated to directly contribute around 626 kg Ce. This level of energy output is equivalent to combustion of 2.5 gal of gasoline for each hour of irrigation pumping.

Average emissions from all management practices for OCC are estimated at 14.15 Mg of C per year, or 0.30 Mg C/ha/year.

13.3.3 Total Carbon Sequestration Potential of Turfgrasses

Based on sequestration and emissions through HCC at OCC, course emissions exceed sequestration in about 30.4 years after course construction (Table 13.4). Initial sequestration rates are significantly higher than emissions rates by HCC with golf turfgrasses having initial large C sequestration potential. Over time, however, these soils sequester less and less C as they reach an equilibrium point while the emissions through maintenance practices remain stable, thus diminishing the potential of SOC sequestration (Table 13.4). Taking into account both sequestration and HCC of maintenance, OCC had a net sequestration potential of 1,611 Mg C over 91.4 years. Estimated sequestration potential for all Ohio golf courses (1.04 Tg C) and all U.S. courses (28.7 Tg C), also showed significant net C sequestration potential (Table 13.5). However, due to the intense maintenance emissions all C sequestered at OCC is negated by emissions in 205 years after the course construction. Similarly, all Ohio golf course sequestration is negated by emissions after 216 years and all U.S. course sequestration is negated in 269 years after the course establishment (Table 13.6).

Table 13.4 Total carbon sequestration, emission and potential for Oakhurst Country Club (47.66 ha) expressed per year, and over a 91.4 year period

Years since turfgrass development	C seq estimated rate (Mg C/year)	C emissions rate	C seq estimated potential
0.0–12.0	131.8	14.2	117.6
12.0–14.0	74.9	14.2	60.8
14.0–24.0	46.7	14.2	32.5
24.0–30.4	20.3	14.2	6.1
30.4–62.0	13.2	14.2	−1.0
62.0–68.2	11.5	14.2	−2.7
68.2–81.0	4.2	14.2	−9.9
81.0–91.4	3.4	14.2	−10.8
91.4–	0.0	14.2	−14.2

Years since turfgrass development	C seq/time period (Mg C)	C emissions/time period	C seq potential/time period
0.0–12.0	1581.4	169.8	1411.6
12.0–14.0	149.8	28.3	121.5
14.0–24.0	466.6	141.5	325.1
24.0–30.4	129.9	90.6	39.3
30.4–62.0	416.2	447.1	−31.0
62.0–68.2	71.3	87.7	−16.4
68.2–81.0	53.9	181.1	−127.2
81.0–91.4	34.9	147.2	−112.2
91.4–	0.0	79.2	−79.2
0–91.4	2904.0	1293.3	1610.7

Table 13.5 Total time necessary from golf course establishment for course carbon sequestration to equal course carbon emissions

	C seq rate	Total C emissions	Total C seq potential
Oakhurst Country Club	2904.0 Mg	1293.3 Mg	1610.7 Mg
All Ohio golf courses	1.81 Tg	0.77 Tg	1.04 Tg
All U.S. golf courses	43.1 Tg	14.4 Tg	28.7 Tg

Table 13.6 Total time necessary from golf course establishment for course carbon sequestration to equal course carbon emissions

	C seq potential over 91.4 years (Tg C)	C emissions/year after equilibrium (Mg C)	Total time until C emissions = estimated carbon seq (Years)
Oakhurst Country Club	0.002	14.15	205
All Ohio golf courses	1.04	8378.93	216
All U.S. golf courses	28.7	1.58 × 105	269

13.4 Conclusions

Conversion to golf turfgrass systems can sequester SOC over time. However, the estimates of technical potential must be viewed in light of numerous factors. One, the net C sequestration potentials reported in this chapter are for newly constructed courses. Thus, the actual SOC sequestration rates and pools may be lower than estimations reported herein as some courses are over 100 years old and may have reached the equilibrium values. Two, emissions due to HCC continue to accrue even after courses have reached the equilibrium. Thus the net sequestration continues to diminish over time. Three, all courses reported in this study were constructed on agricultural soils, many of the nations courses are in fact built by conversion of forests or wetlands. In such cases, the actual clearing and draining of land may have higher HCC than any subsequent turf sequestration before course construction is even finished. So while all courses may show the ability to sequester C over time, the C intensive construction and maintenance practices used to install and maintain courses in pristine shape may not only limit the global climate mitigation potential of these soils, but also exacerbate C emissions once the initial SOC sequestration rate has ceased.

Although golf turfs use a range of C-intensive maintenance practices, HCC can be reduced with proper construction and use of BMPs thereby enhancing the potential to efficiently and effectively sequester a significant amount of atmospheric CO_2. While new courses are still in the planning process, all efforts should be made to choose a previously disturbed and degraded site. Forest clearing and wetland draining must be avoided at all cost to maximize the net sequestration potential of the future course. In addition, each course must also develop and implement an environmental management plan which not only minimizes damage to the environment but also lessons the courses C footprint. Growing grasses which are drought resistant, hardy, slow growing, and fungi and pest resistant can drastically reduce the HCC of intensive practices necessary to maintain a healthy turf.

In addition to prudent management decisions in the planning stages, there are a host of steps that course managers can undertake to maximize the sequestration potential of courses and minimize HCC of turf development. In order to ensure proper BMPs that limit unnecessary C-intensive maintenance practices, course managers should be familiarized with such techniques. A list of suggested BMPs to reduce the HCC of course maintenance and increase the overall net potential of golf turfgrass systems to sequester C can be seen in Table 13.7. Some of these BMPs include: limiting mowing in summer months when grass growth is slowed, watering at cooler times of the day to reduce losses by evapotranspiration, and watering only when is absolutely necessary. Excessive mowing during hot summer months when grass growth is limited leads to the unnecessary combustion of large amounts of C-intensive fossil fuels. While many courses mow once or even twice daily per routine, a brief daily check for necessity could save countless gallons of fuel and increase the net SOC sequestration potential of the course. Similarly, many courses water during the hottest hours of the day, which leads to a higher net water and

Table 13.7 Suggestions to minimize C emissions due to turfgrass maintenance and maximize golf course global climate change mitigation potential

Activity	Do's	Dont's
Irrigation	• Water between 12 and 4 am to minimize evaporation • Water only when necessary • Fix leaky/broken sprinkler heads and pipes promptly to avoid water loss and fungal infections	• Water during the afternoon when temp and evaporation are high • Use automatic timers that water even when it is raining • Allow leaky sprinklers to ooze water, which may lead to increased fungicide use and water loss
Fertilizer and pesticide use	• Apply only when necessary • Apply by hand when able • Spot application for localized infections • Buy products produced close to home to reduce transport emissions	• Apply routinely regardless of necessity • Use fuel driven equipment for application when unnecessary • Buy products from out of state or region leading to high transport emissions
Fuel use	• Mow greens a max of once/day • Mow fairways and rough only when needed • Use new energy efficient hybrid mowers • Use battery powered golf carts	• Mow greens twice daily • Mow fairways and rough routinely, even during months with minimal growth • Use diesel run high fuel burning equipment • Use gas powered golf carts
Other	• Utilize grasses resistant to local fungi and pests, ones that are drought resistant, or those that are slow growing • Construct "green" clubhouses • Educate course managers • Institute environmental management plans • Choose sites that will limit emissions and destruction of natural land use	• Use grasses which require lots of fertilizer/pesticide, water and mowing even when avoidable • Use out of date inefficient maintenance techniques • Ignore environmental impacts of course construction • Choose sites requiring major land use alterations and habitat destruction

energy usage as much of the water is evaporated prior to infiltration. Furthermore, many courses utilize automatic timers that trigger irrigation even during rain events. Such practices greatly exacerbate the HCC and reduce the net potential of courses to sequester C.

Additional BMPs include utilizing spot application of fertilizers and pesticides (precision management) ensuring the proper amount of use at each location. This will not only lead to less chemical use but will also prevent excess chemicals from leaching into groundwater or running off into nearby streams, lakes or rivers. Applying chemicals by hand when possible will also lower the HCC associated with turfgrass maintenance, again allowing for more precise application and the avoidance of waste. When available, the purchasing of local products can also reduce HCC by limiting transport emissions and may also reduce the price of shipping over extended distances. Additionally, as technology continues to advance, hybrid mowers can replace broken equipment. Solar and wind technology may be used to run golf carts, clubhouses, course vending machines, and pump houses. All of these BMPs should be utilized when possible to ensure that the HCC of maintenance emissions are limited and the SOC sequestration potential of the courses are maximized. In addition to the environmental benefits of these activities, they may also lead to significant financial gains for course owners and operators. By limiting the amount of pesticides, fertilizers, fossil fuels, and water used to maintain a course, the operating cost of the course will also decrease as less product will need to be purchased and less manual labor will be necessary to complete maintenance activities.

Finally, alternate turfgrass systems such as home lawns and recreational fields may hold a greater potential to offset emissions because HCC of these systems are often lower than those of golf courses. Home lawns are generally not mowed and irrigated daily, and receive far less applications of fertilizer and pesticides. In some areas of the country turf may require no outside irrigation, supplemental fertilizers, pesticides. These regions would make ideal locations for turf construction and may create ecosystems with a high C sequestration potential. If sequestration rates in these alternate types of turfgrasses are equivalent to those in golf turfgrasses they may prove to be an important resource for C sequestration.

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Chapter 14
Modeling Carbon Sequestration in the U.S. Residential Landscape

G. Zirkle, Rattan Lal, Bruce Augustin, and R. Follett

Abstract The role of terrestrial carbon (C) sequestration in urban areas is an issue of increasing interest. Therefore, a model was developed to evaluate the potential of sequestration in U.S. residential landscapes. The model contrasted C sequestered by trees, shrubs, lawns, and a forest. The first objective of this model is to document the typical urban landscape in terms of lawn area, number of trees and shrubs, area of landscape and garden beds, hard surfaces, and buildings. The second objective is to estimate the annual rate of C sequestration of the residential landscape based on the percentage of lawns, trees, and shrubs. Urbanized land occupies approximately 40.6 million hectares (Mha) with an average of 41% of this land under residential use. Tree, shrub, and lawn C sequestration rates were estimated based on the typical US residential lot size of 2,000 m^2. A typical US home is 93 m^2 with a 2-car garage or carport size of 38 m^2 and a deck or patio of 38 m^2. The house is generally sited in the middle of the lot, with a driveway of 168 m^2 and a sidewalk of 122 m^2 along the front of the lot, leaving a landscape area of 1,541 m^2 Two landscape regimes were modeled, the first involving a minimal area with one landscape bed in the front of the house 13 m long containing 5–10 shrubs approximately 0.6–1.2 m in length and width, 2 trees, and a minimally managed lawn. The second model had the maximum area with several landscape beds 43 m long surrounding the perimeter of the house containing 17–35 shrubs approximately 0.6–1.2 m in length and width, 6 trees, and

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a highly maintained lawn. The total or gross C sequestration rate is 3.4–5.9 kg C tree^{-1} year^{-1} and 0.07–0.23 kg C shrub^{-1} year^{-1}. The C sequestration rate for lawn is 254–2,043 kg C ha^{-1} year^{-1} based on the models described above. A minimal landscape sequesters an average of 25–116 kg C ha^{-1} year^{-1} (39–178 kg C yard^{-1} year^{-1}). Based on the model, residential lawns and landscapes fall within the same range of CO_2 sequestration as a forest land. In a minimal landscape, approximately 8.6% of the C is sequestered by trees, 1.3% by shrubs, and 90.1% by the lawn. The maximum landscape model estimates C sequestration at an average rate of 42–191 kg C ha^{-1} year^{-1} (65–294 kg C yard^{-1} year^{-1}) comparising 15.5% by trees, 2.7% by shrubs, and 81.8% by the lawn.

Keywords Soil • C sequestration • Global warming • Lawn management • Turfgrass • Residential yard • Hidden C costs

List of Abbreviations

BMP	Best Management Practices
C	Carbon
DIY	Do-it-Yourself
HCC	Hidden Carbon Costs
LCA	Lifecycle Analysis
MI	Minimal Input
NPP	Net Primary Productivity
SOC	Soil Organic Carbon

14.1 Introduction

Urbanized land occupies 3.5–4.9% of U.S. land area comprising a total of approximately 40.6 million hectares (Mha) (NAR 2001; Nowak et al. 2001). An average of 41% of urban area is under residential use (Nowak et al. 1996, 2001). Being a major land use, residential land must be evaluated for its impact on the regional and national C cycle.

Atmospheric CO_2 is photosynthesized and stored as plant biomass. Some of the biomass-C is humified and stored long term in the soil as soil organic carbon (SOC). The SOC pool increases when humus content increases and gains of biomass C into the soil exceed losses (Bruce et al. 1999). The net C gains change over time as plants and soils change with input, senescence, and decay. Land use and plant and soil management practices influence the rate of C sequestration. Estimated C density in U.S. urban soils is 7.7 ± 0.2 kg m-2 (Pouyat et al. 2006).

Trees and shrubs remove significant amounts of air pollutants and their presence is important to improving environmental air quality. The pollutants removed include ozone (O_3), particulate matter less than ten microns (PM_{10}), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), and carbon monoxide (CO) (USDA 2006). Urban biota are estimated to remove 711 Tg (Teragrams = 1 million metric tons) of pollutants from the air in the US alone (Nowak et al. 2006). Estimates of the storage of C and gross C sequestration by urban trees in U.S. cities range from 1.2 Tg in New York City to 19.3 Tg in Jersey City (Nowak and Crane 2002). Gross C sequestration for urban ecosystems of the U.S. is estimated at 22.8 Tg C (Nowak and Crane 2002). The amount of C sequestered in trees is estimated at 1.4–54.5 kg C tree^{-1} year^{-1} (Nowak and Crane 2002; England et al. 2006).

Grass lawns in the U.S. occupy 6.4 Mha of area with an average home lawn size of 0.08 ha (Vinlove and Torla 1995; NAR 2001; Zirkle et al. 2011). Lawns have the potential to sequester C at rates as high as 1.0 Mg C ha^{-1} year^{-1} (Gebhart et al. 1994; Contant et al. 2001; Qian and Follet 2002, 2010). A net C sequestration rate in household lawns is estimated at 254–2,043 kg C ha^{-1} year^{-1} (Zirkle et al. 2011), based on low to high management regimes and considering hidden C costs (HCC) of mowing, irrigation, fertilizer, and pesticide applications.

The rate of C sequestration over time can be measured directly and indirectly (Smith et al. 1993, 2008; Bruce et al. 1999; Rickman et al. 2001). Mathematical C sequestration modeling is well developed and used to predict C cycles under various environmental and regional scales (Smith et al. 1993, 2008; Bruce et al. 1999; Lal 2004; Post et al. 2004).

While home lawns, along with trees and shrubs, have potential to sequester C, research data on SOC dynamics for residential yards is scanty. Thus, the objective of this study was to predict the rate of terrestrial C sequestration by trees and shrubs, including the SOC sequestration in soils under household lawns in the U.S. Specific objectives of this study were to: (i) document the typical urban landscape in terms of lawn area, number of trees and shrubs, area of landscape and garden beds, hard surface areas, and building areas, (ii) estimate the annual rate of C sequestration of the typical home landscape, including the percentage lawn, trees, and shrubs, to contribute C sequestration, and (iii) evaluate the range of annual U.S. C sequestration rates including those by trees, shrubs, and lawns.

14.2 Materials and Methods

14.2.1 Residential Landscape Area

This model estimates the SOC sequestration under turfgrass and C stock in tree and shrub biomass of a typical residential yard in the US. Available data from literature were used to configure and model C sequestration rates for a typical residential home lot.

Modeling, based on life cycle analysis (LCA), was used in this study to estimate C sequestration potential for residential lots over a national scale. Net C sequestration was comprised of net lawn SOC sequestration plus that sequestered by tree and shrub biomass C.

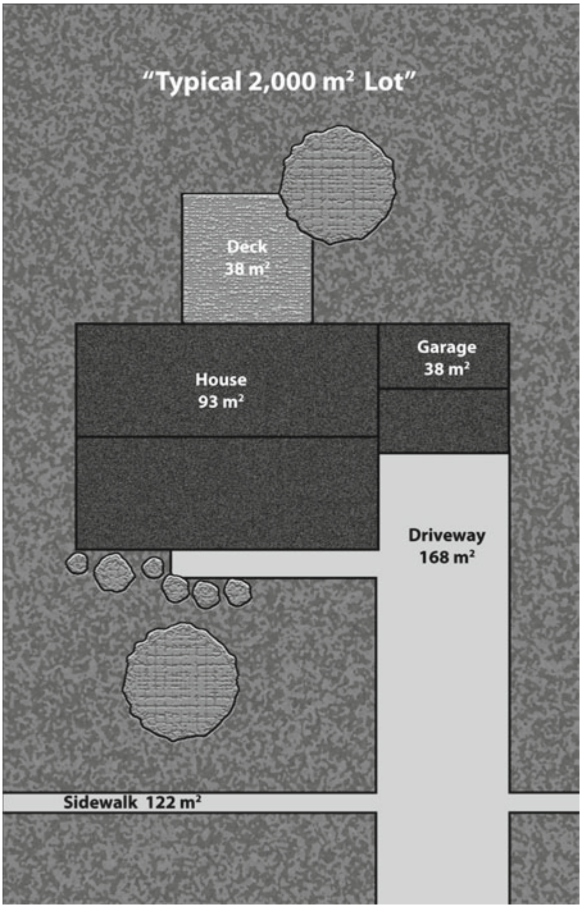

Fig. 14.1 Minimal landscape lot

The average U.S. single family residential home lot of 2,000 m^2 is comprised of an average house size of 93 m^2 (USCB 2008). The majority of homes have a 2-car garage or carport of 38 m^2 (USCB 2008). A deck or patio is common for the average residential home and is 38 m^2. The house is assumed to be in the middle of the lot with a driveway of 168 m^2 and a sidewalk of 122 m^2. The remaining 1,541 m^2 is the landscape area.

Since residential home landscapes differ in landscape-bed size and plant species diversity, a model was developed by using a range of landscape-bed sizes. The minimum is based on a home with one landscape-bed, and 13 linear m long in the front of the house (Fig. 14.1). The bed length is determined from a house with 40 linear m along the front of the house and leaving 1/3 open for a walkway and entryway. The maximum includes a home with 43 linear m of landscape-beds surrounding the perimeter of the house (Fig. 14.2). This bed length was also determined by leaving 1/3 open for deck, garage, walkways, and entry ways. The range of possible landscape-beds is 13–43 linear m.

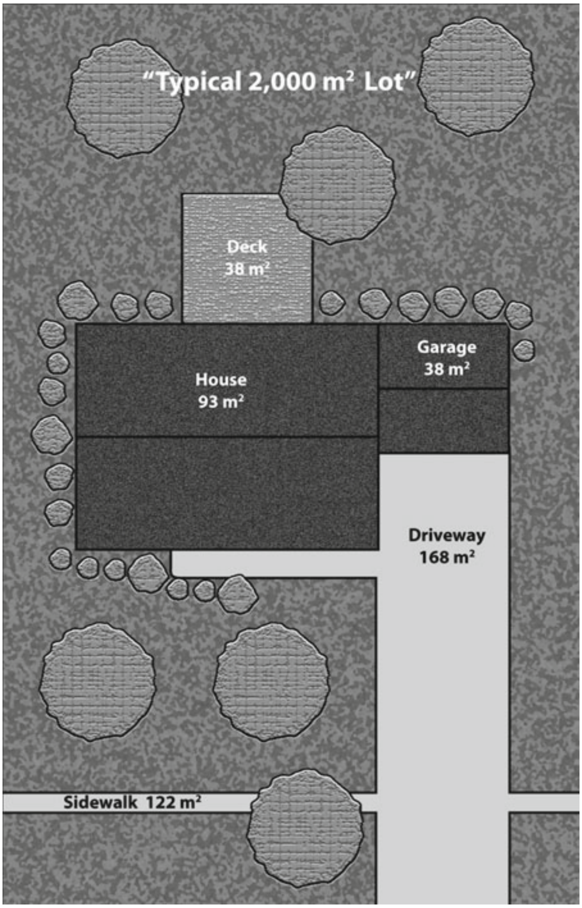

Fig. 14.2 Maximum landscape lot

A large number of tree species and sizes occur in a residential landscape. Therefore, a range in the average tree count yard^{-1} was derived from the available literature. The density ranges from 2 to 6 trees yard^{-1} (Simpson and McPherson 1998; Martin et al. 2003; Swiecki and Bernhardt 2006; Casey Trees 2008). The data in Table 14.1 represents the average tree canopy diameter ranging from 8 to 11 m and was determined from the 10 most common tree species grown in the U.S. (Little 1979; Dirr 1988). The average tree canopy area of 50–133 m^2 was calculated from the average tree canopy diameter.

Shrubs within these beds also range in plant size and species diversity. Spacing of shrubs depends on species and personal preference of density. To keep within typical home residential landscape planting practices, an average range of shrubs yard^{-1} was determined. Shrubs were assumed to be between 0.6 and 1.2 m in length and width, as well as spaced 0.6–1.2 m apart. The minimum landscape bed of 13 linear m contains 5–10 shrubs. The maximum landscape bed of 43 linear m contains 17–35 shrubs. Thus, the range is from 5 to 35 shrubs yard^{-1}.

Table 14.1 Top ten tree species in the U.S. and average canopy sizes (Dirr 1988)

Tree species	Minimal canopy diameter (m)	Maximum canopy diameter(m)
Red maple (*Acer rubrum*)	7	9
Loblolly pine (*Pinus taeda*)	9	11
Sweetgum (*Liquidambar styraciflua*)	11	15
Douglas fir (*Pseudotsuga menziesii*)	5	8
Quaking aspen (*Populus tremuloides*)	6	9
Sugar maple (*Acer saccharum*)	12	15
Balsam fir (*Abies balsamea*)	6	8
Flowering dogwood (*Cornus florida*)	5	6
Lodgepole pine (*Pinus contorta*)	3	4
White oak (*Quercus alba*)	18	24
Average	8	11

Table 14.2 Estimated annual net urban tree carbon storage per year (From Nowak and Crane 2002)

City	Net C ha^{-1} (kg)	Trees ha^{-1}	Net C $tree^{-1}$ (kg)
Atlanta, GA	940	276	3.41
Baltimore, MD	520	136	3.82
Syracuse, NY	540	137	3.94
Boston, MA	490	83	5.90
New York, NY	260	65	4.00
Philadelphia, PA	310	62	5.00
Jersey City, NJ	150	36	4.17

Lawn area was determined by subtracting the amount of canopy area from the trees and shrubs. It was assumed that the area of land under trees and shrubs does not receive adequate sunlight for turfgrass growth.

14.2.2 Tree, Shrub, and Lawn C Sequestration

Average C stored in above and belowground biomass of trees and shrubs was compiled from the literature. Total or gross C sequestration for trees and shrubs was estimated based on individual tree and shrub count.

Total net tree C sequestration ranged from 3.4 to 5.9 kg C $tree^{-1}$ $year^{-1}$(Nowak and Crane 2002) (Table 14.2). This range was calculated using equations reported in the literature (Nowak 1994; Nowak et al. 2001). These data on above-ground biomass were converted to whole tree biomass based on a root: shoot ratio of 0.26 (Cairns et al. 1997).

The tree C sequestration rate calculated from Eq. 14.1 indicates the minimum number of trees sequestered 6.8–11.8 kg C $yard^{-1}$ $year^{-1}$. The maximum number of trees sequestered 20.4–35.4 kg C yard^{-1} year^{-1}. Thus, total C sequestered by trees ranges from 6.8 to 35.4 kg C yard^{-1} year^{-1}.

Net Tree C sequestration rate

$$\begin{aligned}&\text{Minimum tree net C sequestration} = 3.4\ \text{kgC} \times \text{number of trees}\\&\text{Maximum tree net C sequestration} = 5.9\ \text{kgC} \times \text{number of trees}\end{aligned} \tag{14.1}$$

The rate of C sequestration in shrubs was calculated by taking the percentage of canopy area covered by shrubs compared to that by trees. Shrubs 0.6–1.2 m wide have a canopy area of between 0.9 and 3.8 m^2. Shrubs had approximately 0.02–0.04% area and sequestration rate to those of trees. Thus, the C sequestration rate is 0.07–0.24 kg C shrub^{-1}, was calculated from Eq. 14.2 and indicates the minimum rate by shrubs of 0.4–2.4 kg C yard^{-1} year^{-1}. The rate of C sequestered by shrubs was 1.2–8.4 kg yard^{-1} year^{-1}. Thus, the total amount of C sequestered by shrubs ranged from 0.4 to 8.4 kg C yard^{-1} year^{-1}.

Net Shrub C sequestration rate

$$\begin{aligned}&\text{Minimum shrub net C sequestration} = 0.07\ \text{kgC} \times \text{number of shrubs}\\&\text{Maximum shrub net C sequestration} = 0.23\ \text{kgC} \times \text{number of shrubs}\end{aligned} \tag{14.2}$$

Lawn C sequestration rates were calculated by subtracting the hidden carbon costs (HCC) of mowing, irrigation, fertilization, and pesticide use from the gross SOC sequestration (Zirkle et al. 2011). The latter involved a combination of SOC sequestration rates and gross primary productivity (GPP) data available from the literature. The HCC were calculated by converting energy use requirements for lawn management practices into C equivalents (Lal 2003). Home lawns have been shown to sequester SOC at a rate of 254–2,043 kg C ha^{-1} year^{-1}. Minimal managed lawns are mowed once a week without any irrigation, fertilizer, or pest control. Technical potential of C sequestration of these lawns is 254–1,142 kg C ha^{-1} year^{-1}. The rate of C sequestration by the lawns managed at the maximum maintenance level is 517–2,043 kg C ha^{-1} year^{-1}, and includes mowing, irrigation, fertilizer and pesticide applications (Zirkle et al. 2011).

After subtracting the amount of tree and shrub canopy area, yards in the minimal landscape regime have a lawn area of 1,237–1,436 m^2. The lawn C sequestration rate under minimal management is 31–164 kg C lawn^{-1} year^{-1} which is derived from taking the percentage from one hectare of lawn. The maximum landscape has a lawn area of 610–1,226 m^2 and a C sequestration rate of 43–251 kg C lawn^{-1} year^{-1}.

14.3 Results and Discussion

The C sequestration rates for individual tree, shrub, and lawn areas were totaled to attain the gross C sequestration rate for a typical residential landscape in the U.S. Tree count was based on the data available from literature. The area of landscape beds and shrub canopy determines the total amount of shrubs. Two different landscape

Table 14.3 Annual net carbon sequestration rate per yard

Parameter	Trees	Shrubs	Lawns
Minimum C sequestered (kg)	6.8–11.8 (9.3)	0.4–2.4 (1.4)	31.4–164.0 (97.7)
Percent C in each category	8.6	1.3	90.1
Maximum C sequestered (kg)	20.4–35.4 (27.9)	1.2–8.4 (4.8)	43.3–250.4 (146.9)
Percent C in each category	15.5	2.7	81.8

regimes are configured to provide a range of minimal to maximum amount of tree and shrub count yard^{-1} year^{-1}. The minimal residential landscape has a landscape-bed 13 m in length with 2 trees and 5–10 shrubs. The maximum residential landscape has a landscape bed 43 m long with 6 trees and 17–35 shrubs.

A minimal landscape sequesters 39–178 kg C yard^{-1} year^{-1} (248–1,156 kg C ha^{-1} year^{-1}), comprising an average of 8.6% sequestered by trees, 1.3% by shrubs, and 90.1% by the lawn (Table 14.3). The maximum landscape sequesters 65–294 kg C yard^{-1} year^{-1} (421–1,909 kg ha^{-1} year^{-1}) comprising an average of 15.5% by trees, 2.7% by shrubs, and 81.8% by the lawn (Table 14.3).

There are approximately 80 million U.S. single family detached homes (NGA 2004; Augustin 2007). Thus, total national potential ranges from 3.1 to 23.5 Tg C year^{-1} (0.2–2.0 Mg C ha^{-1} year^{-1}). The number of trees and shrubs for U.S. yards ranges from 0.4–2.8 billion shrubs and 160–480 million trees. These ranges are calculated from the assumption that all yards are being maintained at a minimal to maximum landscape. Since yards are managed at various maintenance regimes, this range should fall within the actual range for the U.S.

The tree C sequestration rates were based on the estimations from cities mostly located within the northeastern U.S. Actual C sequestration rates vary among tree and shrub species (Woodbury et al. 2007). USDA (2006) reported that states out of the 109 species studied, the tulip tree (*Liriodendron tulipifera*) is estimated to sequester the most C in Washington, DC, while the American sycamore (*Platanus occidentalis*) sequesters the least amount of C annually. The rates of C sequestration also vary based on tree and shrub size and diameter. As trees and shrubs grow, C content accumulates in the plant tissue. Therefore, larger trees sequester more C than smaller trees due to the difference in plant tissue. When a tree dies and decays, stored C is released through decomposition.

Regional ecosystem conditions and species tolerance also influence the rate of C sequestration and growth among trees, shrubs, and turfgrass ecosystems (Woodbury et al. 2007; Zirkle et al. 2011). For example, areas in parts of the arid southwest may manage yards using the xeriscape strategy (Photo 14.1). Xeriscaped yards include water-efficient landscaping techniques and drought tolerant plant species (Welch 1991). These yards typically include little to no turfgrass areas (Welch 1991), and are mulched with gravels.

Another influence on C sequestration is plant species from a residential neighborhood scale. Interactions between residents within a neighborhood can influence the type of species and landscape regime a resident may choose (Jim 1993). These

Photo 14.1 Example of xeriscape (Courtesy of A. Selhorst)

choices can be driven through observation of what neighbors have done with the landscape or as a result of neighbors' suggestions (Routaboule et al. 1995).

Other herbaceous plants (e.g., perennials and annuals) are found in many residential landscapes. The C balances for these plants are not evaluated in the present study because of the considerable differences in number of species, plant number, and maintenance practices when comparing residential yards across the U.S. Jo and McPherson (1995) documented the herbaceous plants in residential landscapes retain relatively small amounts of C compared to that in trees and shrubs.

Converting C into CO_2 by multiplying the atomic weight of CO_2 (44) and dividing by the atomic weight of C (12), the total technical potential of C sequestration is 11.4–86.2 Tg CO_2 year^{-1} for all residential yards in the U.S. (0.7–7.3 Mg CO_2 ha^{-1} year^{-1}). The potential of minimal landscape ranges from 141 to 653 kg CO_2 yard^{-1} year^{-1} (916–4,240 kg CO_2 ha^{-1} year^{-1}), while that of the maximum landscape from 238 to 1,079 kg CO_2 yard^{-1} year^{-1} (1,545–7,000 kg CO_2 ha^{-1} year^{-1}). These conversions do not include the other pollutants which trees, shrubs, and grasses also remove.

The USDA (2006) documents forest CO_2 uptake at 3,201 kg ha^{-1} year^{-1}. Lawn CO_2 uptake is calculated at 931–4,187 kg ha^{-1} year^{-1} for minimal input (MI) lawns, 2,955–6,710 kg ha^{-1} year^{-1} for do-it-yourself (DIY) lawns, and 1,896–7,491 kg ha^{-1} year^{-1} for lawns maintained at university best management practices (BMPs) (Zirkle et al. 2011). Table 14.4 shows a comparison of lawn CO_2 uptake by, landscape, and forest.

A large fraction of the CO_2 uptake of a forest ecosystem comes from the above ground growth or the wood of the tree (USEPA 2004). When a forest ecosystem

Table 14.4 Comparisons of annual lawn, landscape, and forest CO_2 sequestration rates

Parameter	kg CO_2 ha^{-1} $year^{-1}$
Minimal input lawn SOC sequestration	931–4,187
Do-it-yourself lawn SOC sequestration	2,955–6,710
Best management practices lawn SOC sequestration	1,896–7,491
Minimal landscape C stock	909–4,240
Maximum landscape C stock	1,544–7,000
Forest C stock	3,201

experiences natural tree loss, the CO_2 stored is released back into the atmosphere through decomposition. In a lawn, most of the CO_2 is sequestered below ground in the soil. Lawns experience minimal soil disturbances; therefore CO_2 uptake can occur over a long period of time. Loss of CO_2 from a lawn covered soil occurs through soil and plant respiration, but is typically at minimal rates when compared to the amount sequestered.

The BMP Lawn Management Regime can sequester the most CO_2 at the high range compared to the other regimes in this study. This is closely followed by the high end of the Maximum Management Regime. These applications can increase the amount of net primary productivity (NPP) and SOC accumulated each year. With trees and shrubs, fertilizer and irrigation applications may occur, but NPP is usually lower when compared to a grass species.

Residential landscapes and green spaces have numerous benefits which improve urban environments. Turfgrasses protect soil and water resources by preventing erosion, reducing dust, and filtering pollutants (Beard and Green 1994). Turfgrass also moderate temperature through heat dissipation (Beard and Green 1994). In general, green plants have a positive impact on psychological health by promoting health, and relieving stress (Ulrich 1984; Hull 1992). Residential landscapes can also increase property values, in turn influencing homeowner's decisions on where to live (Getz et al. 1982; Anderson and Cordell 1988).

14.4 Conclusions

A practical model is described to estimate the net C sequestration of a typical U.S. residential yard. The typical U.S. yard sequesters 39–294 kg C $yard^{-1}$ $year^{-1}$ (141–1,079 kg CO_2 $yard^{-1}$ $year^{-1}$) with a total technical potential of 3.1–23.5 Tg C $year^{-1}$ (11.4–86.2 Tg CO_2 $year^{-1}$) for the U.S.

A comparison of lawn, landscape, and forest CO_2 demonstrates that residential lawns and landscapes can sequester CO_2 at a rate equal to a natural forest stock (Table 14.4). A tree stores most of the C above ground in the wood, where a grass stores most of the C below ground in the soil. Therefore, environmental conditions including light availability, temperature, moisture, soil type and fertility levels may influence the rate of C sequestration in the urban landscape more than the type of plant cover and maintenance practices applied.

This model is a large scale evaluation of the C sequestration potential of existing residential yards. It does not include residential yards under xeriscape nor arid-zone yards wherein lawns are not or only marginally used, nor does it include C sequestration by various arid-zone species that might be used in such yards. It also does not include other greenhouse gas emissions from the soil such as methane and nitrous oxide. It also excludes C equivalents for manufacturing of any equipment used for residential landscape management. Validation of this model through field sampling is needed to determine the extent and limits to which residential yards can sequester C. Further research is needed in these areas to increase the precision of the model.

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Part IV
Current Trends in Urban Ecosystems

Chapter 15
Improving Soil Quality for Urban Agriculture in the North Central U.S.

Josh Beniston and Rattan Lal

Abstract An abundance of vacant land exists in the formerly industrial cities of the north central U.S., which have seen tremendous declines in their populations over the past 50 years. Many cities are looking to utilize this land for functional greenspace to improve the overall quality of life. In Ohio, Cleveland and Youngstown contain >1,500 and >2,500 ha of vacant land, respectively, while larger Detroit, MI is estimated to contain between 2,000 and 10,000 ha. Urban agriculture (UA) has emerged as a land use that can provide food production, economic benefits and enhanced ecosystem services on vacant urban parcels. Early data suggest that urban specialty crop cultivation can be quite productive, yielding 2–7 kg m^{-2}, depending on crop and conditions. Given the inherent variability and impact of heavy disturbance common in urban soils, an objective soil quality assessment is necessary to optimize their use for crop production and functional greenspace. Soils in many vacant lots have undergone heavy disturbances, which can lead to severe degradation of their physical, chemical and biological properties. Key constraints to agricultural productivity faced in urban soils include compaction, low levels of organic matter, altered soil moisture characteristics, and lead contamination. Numerous low cost, small-scale intensive methods of improving urban soils exist including the utilization of organic wastes and by-products. The huge quantities of organic materials produced in urban areas have the potential to be processed into high quality soil amendments. Vacant lots in the North Central US can become a valuable asset for community development and improved food security if their soil quality is adequately addressed.

Keywords Urban agriculture • Urban soils • Soil quality • Vacant urban land

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List of Abbreviations

BD	Bulk Density (of Soil)
GHG	Greenhouse Gas
Pb	Lead
MSW	Municipal Solid Waste
OM	Organic Matter
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
UA	Urban Agriculture

15.1 Introduction

Amidst the tremendous increases in the interest and application of urban agriculture (UA) in U.S, this review attempts to relate studies on UA from other parts of the world and ecological surveys of soils in urban areas to the context of UA in North America, and specifically the shrinking cities of the North Central and Eastern U.S. While there is a wide body of literature on UA from around the world, little research to date has been conducted on UA in the U.S. Similarly, while ecological studies of soils in urban areas are increasing, few have assessed the agronomic potential of urban soils. The principal objective of this review is to describe the management of soil quality for crop production in urban areas of the U.S. Additional objectives are to: (1) describe the increase in urban agriculture in the North Central U.S within the context of vacant urban properties, (2) assess the potential of UA to contribute to food production and food security in the U.S., (3) review background information on urban soils in the U.S., (4) describe the importance and utility of soil quality assessment in managing urban soils for crop production, (5) identify specific soil-related constraints to crop production that may be common in urban areas, and (6) suggest small scale intensive management strategies for improving soil quality in urban agro-ecosystems.

15.2 Shrinking Cities

"*What it (the Home Gardening association) has accomplished cannot be computed in dollars and cents. By beautifying vacant lots and yards in nearly every section of the city, it has greatly increased realty values beside adding beauty to the city. But what is more important, it has made the health of the city better. It has gotten people out of doors to cultivate flower and vegetable gardens who before never ventured into a garden. They live and feel better. The value of the work of the association in this way can hardly be overstated*" (Cleveland Plain Dealer 1907). These words were written just over a century ago about the work of a community gardening organization

Table 15.1 Expansion of vacant land and urban agriculture in shrinking cities

Parameters	Detroit	Cleveland	Youngstown
Population 1950 census	1,849,568[a]	914,808[b]	168,330[b]
Population 1980 census	1,203,339[a]	573,822[b]	115,511[b]
Current population (2007 ACS survey)	837,711[a]	395,310[b]	65,056[b]
% Change in population (1950–2007)	−55%	−57%	−61%
Number of vacant parcels	>60,000[c]	15,000 buildings[d]	23,000[e]
Area of vacant land	>1,900[f]–10,000[g] ha	>1,500 ha[d]	2,838 ha[e]
Urban farms and community gardens[h]	>300[i]	75[j]	15[e]
Market/production-based urban farms[h]	20[c]	25–35[j]	2[e]

[a]Population data for Detroit from U.S. Census (www.census.gov)
[b]Population data for Cleveland and Youngstown from Mallach and Brachman (2010)
[c]Urban agriculture data for Detroit from Michigan State Extension, Wayne County, Urban Horticulture Program (2011, personal communication)
[d]Vacant land data for Cleveland from CUDC (2008)
[e]Vacant land and urban agriculture data for Youngstown from the Youngstown Neighborhood Development Corporation (2010, personal communication)
[f]Estimate of amount of vacant land in public land banks in Detroit from Colasanti and Hamm (2010)
[g]Estimate of total area of vacant land in Detroit from Gallagher (2008)
[h]Denotes urban gardens and farms founded during the past 5 years
[i]Wayne County Extension estimates that there are currently more than 300 community gardens in Detroit
[j]Urban agriculture data for Cleveland from The Ohio State Extension, Cuyahoga County, Urban Agriculture Program (2010, personal communication)

in Cleveland, Ohio, and yet the observations expressed here are completely relevant to changes that are unfolding across the U.S. today. Urban gardens and agriculture are going through a great resurgence and generating tremendous interest. During the past few years, UA has become an important topic to a number of disciplines including city planning, community development, horticulture, and soil science. Communities across the country are attempting to utilize UA projects as a means of improving the health and sustainability of cities and as a means of utilizing vacant urban land, nowhere are these trends more visible than in the so called “shrinking cities” of the Eastern and North Central U.S.

While populations are currently increasing in many urban areas, the collapse of industrial economies and the migration of residential and commercial land uses to suburban areas have left many cities in the North Central U.S. with shrinking populations. Nowhere is this trend more visible than in the “rustbelt” cities, such as Detroit, Cleveland, and Youngstown (Table 15.1). These cities share the historical pattern of being designed for peak populations during the regional industrial boom of the early and mid twentieth century. Many of the industries, such as steel and manufacturing, that employed the region’s population either reduced their work force, or in many cases, relocated their operations internationally during the 1970s–1990s. Since that time, populations in these cities have declined steadily and large areas of vacant land and vacant properties have been left behind.

Photo 15.1 The transformation of vacant properties into UA spaces**.** Photos (**a**) and (**b**) show a vacant house (**a**) and a vacant urban lot where a house was recently demolished (**b**) on the South side of Youngstown, OH. The "Lots of Green" program of the Youngstown Neighborhood Development Corporation works to transform vacant properties into green spaces including community gardens (**c**). Raised beds have been created in the gardens with straw bales and imported organic matter (**d**) and pathways have been mulched heavily with cardboard and wood chips (Photos by Youngstown Neighborhood Development Corporation)

The population decline figures of industrial cities in the north central US are staggering. In Ohio, Cleveland has lost 57% of its 1950 population and the smaller city of Youngstown has witnessed a 61% decline from its 1950 population (Table 15.1). Population decline over just the past 30 years (since 1980) has also been significant with Cleveland's population shrinking by 32% between 1980 and 2007, and Youngstown's population shrinking by 44% during the same period (Mallach and Brachman 2010). The infrastructure and footprint of these cities were designed according to peak population numbers, so large population declines have led to large increases in vacant properties. The current figures on vacant land within city limits in these cities and others in the region are just as striking as the population figures. The city of Cleveland currently has at least 1,500 ha of vacant land within city limits, while Youngstown, OH has over 2,800 ha of vacant land (CUDC 2008). Depending on the estimate, Detroit, MI has an area between 2,000 and 10,000 ha of vacant land within the city (Table 15.1).

An abundance of vacant properties can lead to negative patterns of reduced property values and increased crime in affected areas, so many cities have undertaken initiatives to demolish or deconstruct vacant buildings. A recent report on the subject of shrinking cities in Ohio by the Greater Ohio organization and the Brookings Institute has suggested that given the oversupply of housing and current weak markets, shrinking cities in Ohio and throughout the region will need to develop methods of utilizing vacant land that do not rely on traditional redevelopment models (Mallach and Brachman 2010). The opportunities that vacant lots provide in cities are primarily opportunities to create functional greenspaces.

Cities are now attempting to utilize vacant lots for the social and ecological services that greenspaces can provide. An extensive, multidisciplinary literature review on the benefits of urban greenspaces suggests that well planned greenspaces can improve ecological services and the physical and psychological health of people in cities (Tzoulas et al. 2007). Pocket parks, wildflower meadows, storm-water infiltration basins, community gardens and urban farms are all models that have been utilized and proposed in vacant lots (CUDC 2008). The proximity of these vacant lots to large residential neighborhoods has placed them at the center of the upsurge in UA in the north central U.S. As communities look to find solutions to the problem of vacant land, community gardens and urban farms have emerged as a land use with the potential to provide numerous societal benefits (Smit et al. 1996) (Photo 15.1).

15.3 The Potential for an Urban Agriculture in the Midwest U.S.

UA is undergoing a massive resurgence in the North Central U.S., as a win-win solution to the issue of vacant land. Extension personnel in Cleveland, OH estimate that approximately 75 new urban gardens have come online in the past 5 years (Table 15.1), while larger Detroit now has over 300 community gardens and has become a showcase of UA being implemented at a significant scale. Chicago, Milwaukee, and Pittsburgh are among the other cities in the region where UA is expanding rapidly on vacant land. UA has allowed communities to put vacant land, and the citizens who live in its proximity, to work on the task of producing food for those communities. Agriculture in cities can bring a host of societal benefits. The United Nations Development Programme's (UNPD) review of UA around the globe has suggested that it increases nutrition, enhances food security, and creates employment opportunities (Smit et al. 1996).

15.3.1 Urban Agriculture and Food Security

While little research exists on UA efforts in the US and more developed countries, a large number of studies have documented social and economic benefits of UA in Africa, Latin America, and Asia (Pearson et al. 2010; De Bon et al. 2010).

A recent review suggested that UA is done primarily for recreation in developed economies, while its primary objective is enhancing food security in developing economies (Pearson et al. 2010). Similarly, a common sentiment in this body of research is that UA offers a greater benefit to populations in the global south, as people in the northern cities of the U.S. tend, on the whole, to spend a small percentage of their total income on food, live in cities with better food distribution networks, and have higher quality foods available to them, than people in the cities of the global south (Redwood 2010; Smit et al. 1996). While it is true that conditions of food access are different in the U.S., compared with the global south (Smit et al. 1996), there is strong evidence that UA has a significant role to play in the goal of improving food security in the cities of the U.S.

In a report focused on improving the quality of life in Ohio's cities, researchers from the Brookings Institute and the Greater Ohio organization stated that in Ohio the opportunity exists to implement UA at a scale where it can be a major resource for both community food security and economic development (Mallach and Brachman 2010). These opportunities stem largely from the fact that vacant urban land tends to be centralized among the most economically disenfranchised urban areas (Mallach and Brachman 2010; CUDC 2008). These areas often face serious challenges with regard to food security and access to fresh food: a phenomenon termed "urban food deserts." Food deserts are defined as areas where nutritious foods, such as fruits and vegetables, are simply not available to the average citizen in the quantity needed to make up a healthy diet (Wrigley 2002). Research has demonstrated that low-income, non-white urban areas in the U.S. tend to have higher densities of fast food restaurants (Block et al. 2004), higher densities of liquor stores (La Veist and Wallace 2000), and lower densities of grocery stores (Chung and Myers 1999), compared with more affluent areas. A walk through a low-income neighborhood in any American city is likely to validate these observations, as it is far easier to purchase alcohol, tobacco, and sugar-rich candies than it is to purchase fresh vegetables or healthy groceries in the corner stores of major cities.

The current global economic depression and concurrent rise in food prices appear to be exacerbating food insecurity in urban areas. The most recent USDA survey of food security in the U.S., with data from 2008, indicated that 14.7% of U.S. households experienced some food insecurity and 5.7% of all households had very low food security, during 2008 (Nord et al. 2009). These levels of domestic food insecurity are the highest recorded by the USDA, since they began food security surveys in 1995. There is also evidence that globally the urban poor are the group experiencing the greatest food insecurity impacts from the global financial crisis and increasing food prices, as these populations are highly dependent on the cash economy and purchased foods to meet their dietary needs (Ruel et al. 2010). UA, while far from being a complete solution to this problem, may at least offer urban populations a reliable, affordable food source and an increased access to nutrient rich foods (Ruel et al. 2010; Zezza and Tasciotti 2010). World War II era "Victory Gardens" offer an American historical precedent for implementing UA during times of economic crisis and are still cited among the most celebrated examples of gardening and civic activism.

UA is improving access to nutritious foods in the inner cities of the U.S. Data from Cleveland, OH demonstrate that many urban gardens are located in parts of the city that meet a food access criteria to be considered food deserts (CUDC 2008). A survey of community gardeners in Flint, MI observed that adults who participated in community gardens were 3.5 times more likely to consume 5 daily servings of fruits and vegetables than neighbors who did not participate in community gardens (Alaimo et al. 2008). A survey of UA data from 15 developing nations also demonstrated consistent correlations between participation in UA and adequate dietary nutrition (Zezza and Tasciotti 2010).

By virtue of their smaller, more intensive scale, urban gardens and farms commonly focus on producing high nutrition vegetable and fruit crops (often termed specialty crops) (De Bon et al. 2010). Examination of UA operations in Africa and Asia found that they were primarily producing nutritious vegetable crops, such as leafy greens (De Bon et al. 2010). The perishable nature of these crops, and the scale of urban markets have made specialty crop production a good niche for urban producers globally.

15.3.2 Production Potential of Urban Agriculture

UA produces significant quantities of food in the many cities of the world where it is a widespread practice. A UNDP report on UA estimated that upwards of 15–20% of the world's food supply may be produced by UA (Smit et al. 1996). Mougeot (2005) provided an excellent synthesis of the available data on food production through UA which demonstrated that UA provides a significant dietary contribution in many of the world's cities, including: a number of African cities that produce 90–100% of their leafy vegetables; 60% of food consumption in 25% of poor households in Harare, Zimbabwe; 47% of produce in urban Bulgaria; and 500,000 Mg of produce in urban Poland. In Cuba, UA accounted for around 60% of all vegetable production in the nation during 2001(Premat 2005).

Recent production estimates for American cities suggest that UA can make measurable contributions to specialty crop markets in the U.S. A recent study estimated that Detroit, MI could produce 31% of the vegetables and 17% of the fruits currently consumed by city residents on between approximately 100–350 ha, given either relatively "high" or "low" yields (from Jeavons 1995) from bio-intensive agriculture (Colasanti and Hamm 2010). Researchers from the University of Pennsylvania conducted an extensive survey of 226 community gardens in Philadelphia during the 2008 growing season (Vitiello and Nairn 2009). They estimated that over 900,000 kg of vegetables, worth over $4,000,000, were produced in these gardens and distributed throughout the community through informal networks.

The potential of UA to contribute to food systems in the U.S. is just beginning to be understood. A major gap in current research is data on measured yields of specialty crops in urban areas. Table 15.2 is a compilation of available data on UA production statistics for North America. One significant message that these data

Table 15.2 Measured urban agriculture yields

Source	Location	Date	Agricultural system	Crop	Yield/area (kg m^{-2})
Vitiello et al. (2010)	Camden, NJ USA	2009	Community gardens	Mixed veg	2.5
Vitiello and Nairn (2009)	Philadelphia, PA USA	2008	Community gardens	Mixed veg	6.8
Kovach (2010)	Wooster, OH USA	2006–2009	Urban polyculture experiment	Strawberry	4.7
				Raspberry	6.0
				Tomato	8.8
				Edamame Soy	1.4
				Blueberry	4.3
Companioni et al. (2002)	Cuba	1999	"Organoponicos"	Mixed veg	24
Zandstra and Price (1988)	Michigan USA	Averaged	Commercial agriculture "good" yields for vegetable crops	Beans (snap)	0.9
				Cucumber	2.2
				Tomato	3.4
				Mixed cooking greens	1.3

Table 15.3 Altered environmental factors affecting crop productivity in urban areas (Adapted from Eriksen-Hamel and Danso 2010)

Environmental condition	Alteration in urban areas
Temperature	Generally increased
Air quality	Increased levels of ozone (O_3), nitrous oxides (NO_x), sulfur dioxide (SO_2) and suspended particulate matter (SPM)
Solar radiation	Reduced by air pollution
Climate	Urban heat islands and altered hydrologic cycles

convey is that the UA production numbers are significantly higher than the average commercial yields of vegetables (Zandstra and Price 1988). This is likely due to the small-scale nature of UA operations, as many researchers have noted that small scale systems can be more productive than commercial scale agriculture because small scale systems tend to be more intensive in nature (Pretty 1997). Kovach's (2010) data show the promise of well tended fruit crops to be highly productive in urban spaces. His research on intensive polyculture production has also demonstrated that fruit crops can generate higher economic returns for urban scale producers than vegetables. These early figures and estimates suggest that this is merely the beginning of understanding the potential impact that UA can have on the North American food system.

15.3.3 Agroecology for Urban Agriculture

While the social and economic benefits of UA have been extensively researched, little research exists on the agronomy and ecological management of UA systems (Eriksen-Hamel and Danso 2010). While horticulture and crop production are very similar in nature in both urban and rural areas, there are a number of unique ecological influences in urban areas that affect crop production. Urban landscapes have unique biogeochemical conditions and processes compared with more natural ecosystems and agricultural landscapes (Kaye et al. 2006). Many basic ecosystem components and processes are profoundly altered within cities, including climatic conditions, water infiltration, nutrient cycling, resource inputs, and vegetative cover and composition (Pickett et al. 2001; Kaye et al. 2006). All of these processes have profound effects on agricultural systems, so it follows that agronomic conditions in urban areas are also altered.

Eriksen-Hamel and Danso (2010) identified thematic topics where more research is needed to better understand the effects of urban biophysical conditions on UA and management responses to those altered conditions. They identified both altered ecological conditions affecting crop growth in urban areas (Table 15.3) and limiting factors that may constrain crop production in UA. Their list of potential production constraints includes water availability, nutrient supply, soil degradation, pest pressure and soil pollution. These are all issues which can be addressed, at least to some degree, through the sustainable management of urban soils.

In cities, as in all environments, the understanding and management of soils is a basic starting point for sustainable agriculture. The remainder of this review explores the unique ecology of urban soils, the agronomic assessment of these soils, soil-related constraints to agricultural productivity that are common in urban soils and intensive management strategies aimed at alleviating these constraints. UA represents a frontier for soil and agronomic science. Developing research and management programs focused on the agro-ecological management of urban soils can allow for the more effective utilization of these areas for crop production and improve UA's ability to contribute to food security.

Intrinsic variability, and the fact that urban soils exist on a gradient between strong and poor overall quality suggests that urban agriculturalists must take an analytical, site-based approach when determining agronomic management options for these soils. The overall quality and health of vacant lot soils is strongly determined by their previous land use. Soils on lots that have been under perennial vegetation for long periods of time are likely to be of strong overall quality (Grewal et al. 2010). These soils may be readily suited for horticultural production. Conversely, soils on lots that have recently undergone building demolition are likely to have been heavily degraded by that process and require significant targeted management to improve their overall condition (US EPA 2011). The variability in urban soils, coupled with the high number of soil properties that may be affected by the urban environment, make comprehensive soil quality evaluation an excellent management choice for these soils.

15.4 The Soil Quality Framework

The soil quality framework is a useful tool for assessing site-specific soil conditions and developing adaptive management strategies. The concept of soil quality (or soil health) is generally recognized as "the ability of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality and promote plant, animal and human health"(Doran and Parkin 1994). Healthy soil function promotes the robust functioning of the wider ecosystem. Many essential ecological services are provided by soils, including: hydrological cycling, supporting plant growth, the cycling and storage of plant nutrients, the decomposition of organic matter, and the moderation of biogeochemical cycles (Daily et al. 1997). Deriving these functions from soils may be especially critical in urban areas (Table 15.4), where areas of soil and vegetation must provide ecosystem services within a much larger landscape of impervious surfaces. The terms soil quality and soil health are often used interchangeably. Soil quality is the term that is generally thought of as representing quantitative research on the subject, while soil health represents a farmer's assessment of these properties in the field. Soil health may be a useful term to educators as the connection between healthy soils, healthy crops, and healthy people is concrete and easily explained.

Soil quality is typically evaluated by field and laboratory analyses of a suite of soil physical, chemical, and biological properties. Specific properties are chosen based on their value as 'indicators' of the effects of management on the soil. There is no standard for choosing soil indicators. Ideally, indicators are chosen based

Table 15.4 Critical ecological functions supported by urban soils and methods useful in their assessment

Ecological function	Methods of assessment
Stormwater infiltration	• Infiltration measurement • Saturated hydraulic conductivity
Sorption of pollutants (including heavy metals)	• Contaminant bioavailability assays
Sorption and transformation of excess nutrients	• Regular analysis of nutrients of interest in critical areas
Soil C sequestration	• Measurement of SOC pool
Habitat for micro-organisms and invertebrates	• Microbial biomass and activity assays • Invertebrate assays • Enzyme assays • Total porosity
Foundation for plant growth	• Soil quality assessment

on the unique ecological and management conditions of a given soil system or the goals of a research project. Values measured for individual properties are then typically scored against measured and reported distributions from similar ecological conditions (Andrews et al. 2004; Gugino et al. 2009) and tabulated into an index, which gives a score of overall soil health (Karlen and Stott 1994). Thus a completed soil quality report for a site includes scores for individual soil properties and the overall score of the site's soil quality.

The Cornell Soil Health team has developed a unique approach to evaluating and managing soil quality (Gugino et al. 2009; Schindelbeck et al. 2008). Their soil quality lab presents results in a report card fashion that provides scores (0–100) for the individual indicators and an overall soil health score. Overall scores of 0–30 are considered low, while above 70 are in the optimum range, and those between 30 and 70 are intermediate. Soil properties receiving low scores (<30) are identified as constraints for that specific soil and these properties receive focus in the development of management plans to improve soil health. The Cornell team has developed the soil quality evaluation as a test that is available to producers and the public for a fee. The completed analysis results in a soil quality scorecard and management recommendations specific to the individual soil's condition.

The process of identifying soil-related constraints to agricultural productivity and targeting them with adaptive management plans is an excellent framework for pursuing improved soil quality. A full soil quality evaluation offers far more information about the overall condition of a soil than standard soil nutrient testing. Standard nutrient testing generally only provides information about a soil's chemical condition and overlooks the critical physical and biological makeup of a soil. A soil that appears to be in good condition from a standard soil test may have severe physical and biological constraints such as degraded structure (Schindelbeck et al. 2008). Furthermore, many soil chemical constraints (nutrient deficiencies, non-optimal pH, etc.) are easily alleviated through fertilizer application while physical and biological constraints require longer-term changes in management practices (Magdoff and Van Es 2010).

Soil quality assessment is a useful tool in differentiating the condition of soils and sustainability of management in a wide variety of scenarios, including: soils receiving different nutrient inputs (Glover et al. 2000), managed by different tillage regimes (Wander and Bollero 1999), receiving different levels of crop residues (Moebius-Clune et al. 2008), cultivated under diverse commodity crop rotations (Karlen et al. 2006) and amended/restored mineland soils (Shukla et al. 2004). Thus, results are promising for utilizing soil quality assessment to differentiate between the condition of urban soils.

Restoration ecologists working in cities have suggested that soil quality evaluation offers an excellent index for assessing the success of restoration projects in degraded urban sites (Heneghan et al. 2008). They suggest that a focused assessment and site-specific management is necessary to improve soil quality in uniquely urban soils (Heneghan et al. 2008, Pavao-Zuckerman 2008). Sites with high levels of degradation require strong interventions to be restored to ecological health (Pavao-Zuckerman 2008). As in agricultural soils, targeted management and manipulation of individual physical, chemical or biological properties acting as constraints, such as soil compaction, may aid in the restoration of overall quality in degraded urban soils by improving ecosystem processes that limit the whole system (Heneghan et al. 2008).

In order to best assess the quality of UA soils, it will be necessary to generate more datasets from North American cities focused on agronomic conditions specific to these soils. These data can then generate soil scoring functions that will provide a more accurate tool for assessing soil quality in urban areas. A key gap in the scientific understanding that must be addressed in moving forward in UA is to identify specific soil-related constraints to production common in urban soils. In the absence of agricultural studies on urban soils in the U.S., this article reviews existing ecological and horticultural studies of soils in cities and identifies soil properties which are regularly subject to degradation in urban areas. This chapter begins with a basic overview of unique qualities of urban soils and then presents potential soil constraints as well as management strategies which have shown promise for alleviating these in an attempt to further the discussion of managing urban soil quality for crop production.

Urban soils, particularly vacant lot soils, may be characterized by a range of agronomic constraints (Table 15.5). The most significant agricultural constraints that may occur regularly in degraded urban soils include the physical constraints of low levels of organic matter (OM), compaction and poor structure, and the chemical constraint of contamination. Constraints of degraded urban soils are reviewed and management options for each class of constraints will be presented. The vast majority of UA in the U.S is currently located in small parcels of land (<.5 ha), with many urban gardens being just single urban lots (approx. .05 ha). Small parcels lead to a small-scale form of agriculture, which requires that urban producers adopt highly intensive methods to achieve successful levels of productivity. Thus, this chapter focuses on methods of soil management that are both small scale and intensive in nature (Table 15.6). These management practices are characterized by requiring few external energy inputs, having low costs, and utilizing locally available resources.

Table 15.5 Potential soil-based constraints found in urban soils

Constraint	References
Soil physical properties	
Compaction	Gregory et al. (2006), Scharenbroch et al. (2005), Pit et al. (1999), Jim (1998b)
Decreased water infiltration	Gregory et al. (2006), Pit et al. (1999)
Soil chemical properties	
Lead (heavy metal) contamination	Witzling et al. (2011), Clark et al. (2006), Finster et al. (2004)
Alkaline pH	Jim (1998b)
Soil biological properties	
Low organic matter	Craul (1992)
Decreased microbial activity	Scharenbroch et al. (2005)

Table 15.6 Recommended small scale intensive management strategies for soil-based constraints to production in urban agriculture

Constraint	Management	References
High compaction	– Cover cropping	Wolfe (1997), Clark (2007), Williams and Weil (2004)
	– Subsoiling	
	– Add organic matter	Craul (1992)
	– Raised beds	Pit et al. (1999)
Poor drainage	– Surface draining earthworks	Craul (1999), Lancaster (2007)
	– Subsurface drainpipe	Craul (1999)
Available water capacity	– Compost application	Pit et al. (1999)
	– Rainwater catchment	Lancaster (2006)
	– Water harvesting earthworks	Lancaster (2007)
Low organic matter	– Compost application	Cogger (2005)
	– Biochar application	Hargreaves et al. (2008)
	– Cover cropping	Clark (2007)
	– Reduced tillage	
Lead contamination	– Soil testing	Witzling et al. (2011), Finster et al. (2004)
	– Raised beds	Hynes et al. (2001), Witzling et al. (2011)
	– Mulching/covering bare soil in adjacent areas	Clark et al. (2008)
	– Apply phosphate fertilizers	Ryan et al. (2004)
Low nutrient content	– Soil testing	Clark (2007)
	– Apply rock fertilizers (lime, rock phosphate, green sand, kelp, etc.)	
	– N-fixing cover crops	

15.5 Urban Soil Ecology

An urban ecosystem can be thought of as a "biophysical social complex" (Pickett et al. 2008). Within this complex, social and economic activities alter and drive many biophysical processes. Urban ecosystems also tend to be defined by heterogeneity, as high levels of spatial heterogeneity have been identified for nearly all aspects of the urban ecosystem. Anthropogenic influence and high levels of spatial variability also characterize soils in urban areas (De Kimpe and Morel 2000). Urban soils can be defined at the most basic level as those existing within the boundaries of a city. Lehmann and Stahr (2007) offered a useful distinction in defining urban soils by differentiating between natural and anthropogenic urban soils. The latter are soils that have been heavily affected by human activities, such as housing, industrial production, and disposal activities.

Early references on urban soils tended to apply a generalized view that soils in cities are heavily disturbed and characterized by undesirable properties that made using them for horticultural purposes difficult (Craul 1992). More recent ecological surveys of urban areas suggest that urban soils are highly heterogeneous and occur on a continuum between soils that are identical to the native soil types of the region and highly disturbed anthropogenic soils (Pickett et al. 2008; Pouyat et al. 2003). Spatial variability in urban soils is often dictated by the dramatic societal influence on urban land. Land use has emerged as a principal predictor for variation in soil properties in urban areas. Studies comparing urban soils under different land uses have observed differences in a whole suite of soil properties, including: soil physical and chemical properties (Pouyat et al. 2007), compaction (Gregory et al. 2006), soil organic C (SOC) content (Pouyat et al. 2003, 2006), microbial biomass C and N (Lorenz and Kandeler 2005, 2006), and lead (Pb) contamination (Chaney et al. 1984; Wagner and Langley-Turnbaugh 2008) (Fig. 15.1).

Heterogeneity is also often seen vertically in the profiles of disturbed, anthropogenic urban soils where horizons are not always found in parallel to the soil surface (De Kimpe and Morel 2000). This profile perturbation is the result of past land use practices of dumping fill and refuse and grading topography. This situation can result in unexpected profile distributions of soil properties. Lorenz and Kandeler (2005) documented a number of soils in their survey of urban areas in Stuttgart, Germany with deposits of high concentrations of SOC deep in the soil profile (>1 m). Another study of Pb contamination in Portland, Maine, U.S.A. reported high concentrations of Pb in subsoils of some residential areas (Wagner and Langley-Turnbaugh 2008).

Pickett and Cadenasso (2009) argued that urban influence on soils is so great that Jenny's (1941) model of soil formation must be adjusted significantly to explain pedogenesis in urban areas. They suggest that disturbance, altered resources, and spatial heterogeneity of social and ecological processes make up an additional level of soil forming processes that often supersede the traditional soil forming factors. In the majority of studies on soils in cities, disturbance and land use history have an overarching influence on soil properties. Given their critical influence on urban soils, it is necessary to consider land use history in siting and managing urban farms and ideal sites are ones that have not been subjected to recent heavy disturbances.

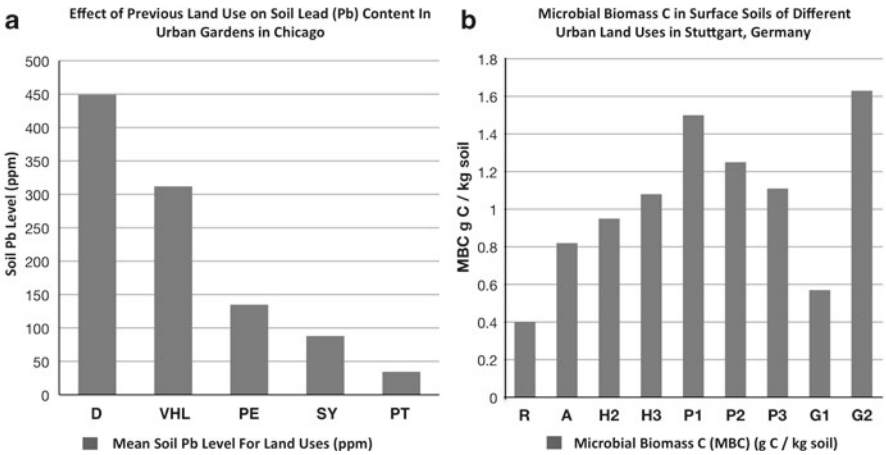

Fig. 15.1 The effect of land use on soil properties. (**a**) Effect of previous land uses on soil Pb in urban gardens in Chicago (Adapted from Witzling et al. 2011). Previous land uses include driveway (*D*), vacant house lot (*VHL*), park entryway (*PE*), schoolyard (*SY*), and park turf area (*PT*). (**b**) Microbial biomass C in surface soils of urban land uses in Stuttgart, Germany (Adapted from Lorenz and Kandeler 2006). Urban land uses include: railway area (*R*), apartment building (*A*), high density urban areas (*H2* and *H3*), public parks (*P1*, *P2*, and *P3*), garden-no vegetation (*G1*) and garden-vegetable (*G2*)

15.6 Soil Compaction

Soil compaction is a common condition in many urban soils (Meuser 2010; Craul 1999), and can be a serious constraint to plant growth and productivity. Soil compaction leads to significant reductions in soil pore space, plant available water capacity, rooting depth, soil biological activity and crop yield (Lal and Shukla 2004). Compaction is generally associated with the heavy traffic and disturbances, processes that occur regularly in the urban environment. Research has documented soil compaction in urban and peri-urban areas due to building construction (Pit et al. 1999), construction vehicle traffic (Gregory et al. 2006), and even for high rates of pedestrian traffic (Millward et al. 2011). Depending on the texture of a given soil, bulk density (BD) values ranging from 1.4 to 1.7 $g\,cm^{-3}$ are thought to have a negative effect on plant root growth, while BD values ranging from 1.5 to 1.8 $g\,cm^{-3}$ are restrictive to root growth (NRCS 2000). Table 15.7 features mean BD values for different land uses reported in studies of soil physical properties in urban areas. A number of soils in those studies demonstrated BD values consistent with serious compaction. The values in the table also represent the variability encountered in urban soil BD, with more disturbed areas having greater BD values.

Soil compaction is a major ecological and horticultural constraint that leads to reduced functioning in affected soils. Compacted urban soils have consistently demonstrated significantly reduced water infiltration rates when compared with adjacent un-compacted soils (Millward et al. 2011; Gregory et al. 2006; Pit et al. 1999).

Table 15.7 High and low bulk density values observed in ecological studies of urban soils

Reference (location)	Bulk density (g cm^{-3})	Land use
Millward et al. (2011)	1.4	Non-naturalized park area
(Toronto ON, Canada)	1.1	Naturalized park area
Pouyat et al. (2007)	1.3	Commercial and industrial land uses
(Baltimore MD, USA)	1.1	Forested areas
Scharenbroch et al. (2005)	1.7	New (<10 years) residential landscape
(Moscow ID, USA)	1.4	Old (>50 years) residential landscape
Stahr et al. (2003)	1.7	Public park
(Stuttgart, Germany)	0.9	2 Sites: public park and garden area
Jim (1998a)	2.1	Heavily used urban park

Bulk density values are from soil surface samples
BD values greater than 1.5 g cm^{-3} may decrease root growth in urban soils (NRCS 2000)

The widespread nature of compaction and the associated reductions in water infiltration in urban areas may lead to reduced soil moisture and lower rates of groundwater recharge (Meuser 2010).

15.6.1 Soil Management to Mitigate Compaction

While soil compaction can be a serious problem, there are several strategies for managing soils to alleviate compaction. Practices designed to alleviate compaction for crop and plant growth include subsoiling and cover cropping, and raised bed cultivation can allow the improved cultivation of crops in a less dense medium above the compacted layer.

Sub-soiling or deep tillage practices have been recommended in the past for alleviating the serious levels of compaction often found in degraded urban soils (Craul 1992). Sub-soiling is an agricultural operation where a chisel plow is used to "rip" or aerate the deeper subsoil layers. It is widely used by farmers to alleviate compaction in agricultural soils. While sub-soiling offers an effective method of reducing soil compaction, the specialized agricultural equipment involved may not lend itself to the smaller lots used in UA. The cost of sub-soiling may also limit its utility in restoring degraded urban soils (Heneghan et al. 2008). Thus, subsoiling is a feasible solution perhaps only for larger-scale urban farming scenarios.

Intensive cover cropping has the potential to be an excellent strategy for urban producers working with compacted soils. Research during the past two decades has documented the ability of a number of deep-rooted cover crops to reduce soil compaction. Species that serve this function exist for most planting times of the year. Oilseed or daikon radish (*Raphanus sativus*) may be particularly well suited for small vegetable operations. It is planted in the late summer/early autumn, winter kills, and has been observed reducing soil compaction in a soybean (*Glycine max*) rotation in Maryland (Williams and Weil 2004). Sorghum/sudangrass (*Sorghum bicolor x S. bicolor var. sudanese*) is a highly productive summer fallow crop which produces tremendous root biomass when it is mowed (Clark 2007). An evaluation of the ability of cover crops to alleviate soil compaction on vegetable farms in upstate New York suggested that sorghum/sudangrass is exceptional in its ability to relieve soil compaction (Wolfe 1997). Other cover crops of interest for compacted urban soils include yellow sweet clover (*Melilotus officinalis*) and annual ryegrass (*Lolium multiflorum*) which, while not tap rooted, can improve soil structure with its extensive root biomass as an overwintering cover crop.

A related soil physical constraint in urban areas is the presence of asphalt. In some neighborhoods, former parking lots, or large asphalt covered lots are the only large areas available for gardening. Removing asphalt is often not economical for UA, so increasingly gardeners in these areas are exploring methods for creating soil on top of the asphalt. Large quantities of compost, wood chips, and other forms of OM are used to create raised beds over the asphalt. Cleveland, Ohio is home to several community gardens and at least two urban farms that grow produce in "soil" created over asphalt (Kroll 2007). Accounts of asphalt gardens are increasing in urban areas across the north central region, yet it is a subject that has not been researched. Questions remain regarding the possibility of contamination from the asphalt and regarding the overall quality of the soil substrate that is created. Given that these soils have been created largely from OM, research is needed on the nutrient content of both the substrate and the resulting crops. Supplemental micronutrient fertilizers are likely necessary to optimize nutrient content.

15.7 Soil Organic Carbon Content

In general, increasing the SOC, or soil organic matter (SOM), content of degraded soils improves their physical condition and soil health. Indeed addition of OM is the primary factor controlling the improvement of soil quality. Most sustainable soil management strategies involve improving the quantity, quality and diversity of OM inputs in a cropping system (Magdoff and Van Es 2010). Increases in a soil's SOC pool are widely known to enhance and improve a wide range of properties and processes in the soil ecosystem, including: increasing invertebrates and microbes, improving soil structure, increasing soil water and nutrient reserves, and improving water quality by sorbing and filtering pollutants (Lal 2007). In degraded agricultural soils, an increase of 1 Mg of SOC has been shown to improve the yield of a number of staple crops (Lal 2006, 2010a, b). The fact that vacant lot soils are often physically degraded implies that they stand to gain significant agronomic benefits from increases in SOC.

15.7.1 Reduced Tillage

The SOC is increased through a variety of soil management practices. SOC regenerative management practices that may be useful for urban producers include: reducing tillage, utilizing crop rotations, growing cover crops, using mulch, and

Photo 15.2 Permanent raised beds are used for vegetable production at the urban farm in Braddock, PA (Photo courtesy of Krista Beniston)

applying compost/manure/biosolids (Lal et al. 1999). Excessive tillage is widely known to reduce SOC, and to degrade soil physical quality. This is type of soil degradation can occur with the excessive use of rototillers, which are widely employed in small-scale vegetable production. Urban producers, and other small-scale farmers, are advised to minimize tillage operations whenever possible, in the interest of building up soil structure and the SOC pool. For those producers operating at a small, garden scale of production, a broad fork may be a useful implement for aerating and working amendments and cover crops into the soil. A broad fork is a large digging style garden fork that is increasingly available through garden equipment suppliers. The broad fork aerates the soil and distributes amendments into the profile without breaking apart the soil structure or inverting the soil surface. Permanent raised beds are another viable option for small-scale producers (Photo 15.2). Raised beds can offer the seedbed and increased soil temperature that tillage provides, but without the degradation of soil structure. They can also be highly productive for small producers, as evidenced in the Cuban "organoponicos" (Table 15.2) (Companioni et al. 2002).

15.7.2 Organic Matter Inputs

Cities are places of abundant organic waste materials. Yard and green waste, food waste, organic industrial byproducts, and municipal biosolids are just a few of

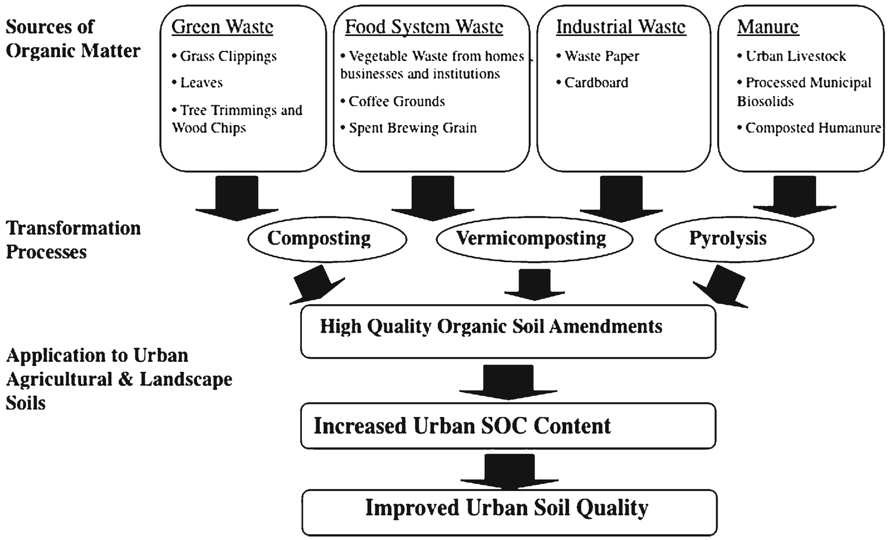

Fig. 15.2 Conceptual model for utilizing organic waste to improve soil quality in urban environments

the many classes of materials which can be used for composting and mulching in an effort to increase the SOC concentration of urban soils (Fig. 15.2). Urban environments offer numerous opportunities to collect both C and N-rich plant materials to combine in the making of stable and nutrient-rich composts. Cogger (2005) reviewed a number of studies on applying green by-product compost to agricultural soils. These studies all showed improvement in a number of soil physical properties, after repeated compost application. These data were used to generate recommendations for disturbed urban landscaping soils: to apply 5–8 cm of compost for plantings in disturbed soils, or up to 25% by volume in extreme examples (Cogger 2005). A recent meta-analysis on the effect of OM applications on urban landscape and tree plantings indicated that soil physical properties and plant growth consistently increased with OM additions to soil (Scharenbroch 2009). In the previously mentioned study of water infiltration in urban soils degraded by construction, heavy compost application increased water infiltration rates between 1.5 and 10 times compared to un-amended soils (Pit et al. 1999).

A number of researchers have investigated the use of municipal solid waste (MSW), which is a compost made up of both green waste and food waste from urban areas. These materials are composted in many municipalities in an effort to minimize materials being sent to the landfill. Hargreaves et al. (2008) reported that the physical, and biological benefits to the soil were numerous with MSW application, but cautioned that the composition and quality of MSW compost materials must be routinely evaluated and that the incorporation of sewage sludge in MSW compost can lead to elevated heavy metal content. In Europe, Diacono and

Montemurro (2010) suggested that long term application of composts, including MSW compost, to cropped soils had many benefits including improvement to a number of soil properties and dramatic crop yield increases, compared with un-amended controls. They reported no evidence of heavy metal contamination from MSW application. Thus, cities and institutions interested in supporting UA systems must begin to explore options for dealing with organic by-products locally through large scale composting programs. These programs can divert material from landfills and improve soil quality by turning waste into a valuable resource (Fig. 15.2).

In addition to producing compost from organic wastes in urban areas, the production of biochar from urban organic waste products is a process that deserves further investigation for improving degraded soils. Biochar is a term used to describe charcoal produced with the goal of creating a soil amendment, energy, or both. Biochar is created through combusting organic materials in the absence of oxygen: a process known as pyrolysis. Pyrolysis provides a number of additional environmental benefits in comparison with composting or natural decomposition. The process produces energy, reduces greenhouse gas (GHG) emissions from decomposition, and leaves behind a greater quantity of stable C in the soil than natural decomposition (Lehmann 2007). Numerous urban green wastes, such a MSW and landscape trimmings, can be used to produce biochar (Lehmann et al. 2006). Pyrolysis technologies range from simple devices that produce only biochar to expensive, electricity generating pyrolyzers.

One of the most significant benefits of biochar is its ability to improve soil quality and crop productivity in degraded soils through increasing their SOC pool (Kimetu et al. 2008; Glaser et al. 2002). While the much of the research on biochar's benefits in soils has been carried out in nutrient-poor tropical soils, results of the early work on biochar in temperate environments suggests that biochar application provides marked improvement to soils in these environments as well (Atkinson et al. 2010). Biochar application has demonstrated improvement to a number of soil physical properties, including increased SOC pool, improved soil structure, increased plant available water capacity, decreased BD, and increased cation exchange capacity (Glaser et al. 2002; Lehmann et al. 2006; Liang et al. 2006; Atkinson et al. 2010).

Biochar application may prove to be a highly effective practice for improving soils for UA in the U.S. for a number of reasons. First, the degraded nature of many vacant lot soils may be greatly enhanced by the addition of stable C, as suggested by the research on degraded tropical soils (Kimetu et al. 2008). Second, cities contain abundant organic wastes (Fig. 15.2). Early research suggests that pyrolysis may be useful for transforming any number of C based materials into biochar (Atkinson et al. 2010). It is rather easy to foresee scenarios where urban organic residues such as food waste, organic industrial byproducts, and landscape byproducts are used to create biochar, and possibly energy, through pyrolysis. The current high costs of pyrolysis technologies and industrially produced biochar suggest that developing methods to produce high quality biochar using low-tech pyrolysis of green waste may be the most appropriate strategy for UA contexts.

15.8 Soil Moisture Characteristics

15.8.1 Poorly Drained Soils

Urban soils can be subject to poor drainage for a number ecological reasons. Constructed physical obstructions such as concrete curbing and uneven site grading may lead to inadequate drainage outlets for a given soil, causing waterlogged conditions (Craul 1999). Drainage may also be inhibited by soil compaction as well as the incorporation of buried impervious materials in disturbed soil profiles. Hydric soils are also common in many cities. These soils formed naturally under wet conditions in wetland or coastal areas. In Cleveland, city planners have mapped the extensive hydric soils, as these soils are less desirable for UA applications (CUDC 2008). The waterlogged conditions in poorly drained soils reduce the growth of most crops.

Management options for poorly drained soils include both surface and subsurface drainage. An evaluation of site-specific conditions is necessary to determine the appropriate drainage options. Surface drainage consists of simple earthworks such as diversion swales that are designed to direct water away from poorly drained areas (Craul 1999). Perforated plastic drainpipe for subsurface drainage is a commonly available landscaping material and most commercial landscapers have experience installing these drains. These simple drainpipes can often be an excellent solution to poorly drained urban soils, but they require at least a slight slope to fully move water from affected areas. Raised bed gardens may also be a solution to utilizing poorly drained soils, though their performance is also likely to be enhanced through the inclusion of other drainage measures on the site.

15.8.2 Soils with Water Deficit

The opposite extreme of a dry moisture regime may also be an issue in urban soils. The profoundly altered hydrology and water tables in urban areas can lead to anthropogenically driven "hydraulic drought" in soils of urban riparian areas (Groffman et al. 2003). These altered environmental conditions, along with the widespread problem of decreased water infiltration in disturbed urban soils, suggest that elevated water deficit for plant growth is a condition likely to be encountered in urban soils.

A variety of options exist for increasing plant available water in UA soils. The addition of OM to the soil in the form of compost, mulch and other amendments can both improve infiltration and increase the water holding capacity of UA soils. Identifying appropriate water sources for irrigation is also a necessity for UA. City water departments can generally install irrigation hydrants for UA projects. These installations may have prohibitive costs in some situations. The harvesting of rainwater for sustainable irrigation has tremendous potential for application in urban areas. Rain barrels that capture water from rooftop gutters have become a common feature in many gardens, but larger tanks and cisterns may be required for market scale UA.

In addition to tanks and barrels, water harvesting earthworks can be an effective means of seasonal irrigation in urban soils (Lancaster 2007). Level swales, infiltration basins, and unique urban applications such as curb cuts with small diversion ditches can all be utilized to capture substantial quantities of rainwater in cities. Lancaster (2006) offers an excellent guide to the principles of designing rainwater catchment systems for multiple scales of users. His book also features case studies of rainwater irrigation of small scale agriculture, including a very inspiring look at the use of rainwater to irrigate edible landscapes and home gardens in the arid urban area of Tucson, AZ.

15.9 Soil Lead (Pb) Contamination

When asked about soil-related concerns, most people working in UA in the North Central U.S., including producers, extension educators, and researchers, mention contamination, and specifically Pb contamination as their primary concern regarding soil quality. Soils in many urban residential areas have been contaminated with Pb from Pb-based paints, exhaust from leaded gasoline and from past industrial manufacturing. Though Pb is no longer used in either paint or gasoline, it was a common ingredient in white paints in the early twentieth century and a standard gasoline additive during much of the latter half of the century. This led to many millions of tons of Pb ending up in American cities in paint and automobile exhaust (Mielke et al. 2011; Mielke 1999). Thus, large quantities of this Pb were deposited in soils, which caused widespread Pb contamination of soils in urban areas. There are trends of increasing soil Pb in soils closer to busy roadways in many cities (Fillipelli et al. 2005).

With Pb no longer being used as an industrial ingredient but having a long half life in soils, the ingestion of soil and airborne soil particulates have emerged as principle Pb ingestion vectors and primary risk pathways for Pb poisoning (Fillipelli and Laidlaw 2010; Laidlaw and Fillipelli 2008; Mielke and Reagan 1998). Pb-contaminated soil can be ingested in a variety of ways, including: direct ingestion of soil by children, incidental soil ingestion by adults working in gardens or green spaces, and ingestion or inhalation of soil-based dust particles on both indoor and outdoor surfaces. Additionally, Pb can be taken up by garden vegetables grown in contaminated soils, at levels capable of causing health concerns (Finster et al. 2004). Thus, even though Pb contamination may not constrain crop growth, public health concerns about soil Pb contamination are high for UA.

15.9.1 Testing Soils for Pb Contamination

Given the potential health risks from Pb exposure, all urban soils should be tested for Pb content before establishing gardens, playgrounds, or UA. Pb contamination is generally most intense in the soil surface (0–5 cm). Thus, surface samples are often used for Pb estimation, but sampling to the depth of rooting (0–40 cm) has also been suggested for garden soils (Clark et al. 2008). High levels of spatial variability of soil Pb concentration have been observed in urban soils (Wagner and Langley-Turnbaugh 2008; Chaney et al. 1984) and in urban gardens (Clark et al. 2006). So, whenever possible, numerous individual or composite samples should be analyzed from a given site, in an effort to capture any variability that may be present.

Currently in the U.S., commercial and university soil testing laboratories use a number of analytical methods to determine total soil Pb content. US EPA methods 3050B (acid digestion) and 3051A (microwave assisted acid digestion) are the protocols used by the US EPA and have been considered the benchmark method for total Pb analysis and are the methods that levels in recommendations are based upon. (Scheckel et al. 2009). Costs of the EPA methods are higher than other extraction procedures though, so these are not always used. Recent comparisons between EPA method 3050B and the more widespread and cheaper Mehlich III soil extraction have indicated strong correlations in results, suggesting that this test which is commonly used for soil nutrient analysis may also be a robust proxy for the EPA methods (Witzling et al. 2011).

Currently there are no standards or regulations regarding total soil Pb levels and food production. The US EPA has suggested that soils with >400 ppm total Pb content require remediation for use as playgrounds (US EPA 2001), so many people use this as a rule of thumb for urban gardens though opinions differ. Research and extension literature on Pb levels and gardens has suggested a range between 100 and 1,000 ppm as a safe upper limit of Pb in garden soils (Witzling et al. 2011) and standards among 10 other nations range from 50 to 450 ppm (Meuser 2010). Certainly, the potential public helath risk associated with sites with greater than 1,000 ppm Pb content makes them unsuitable for UA applications.

Proper analysis of Pb levels in urban soils is a key step in utilizing them for UA, so that an accurate estimate of risk can be attained to guide management strategies. Many sites may have Pb levels that do not require extensive remediation. Recently, researchers at The Ohio State University analyzed soil Pb levels for more than 50 potential UA sites in Cleveland and found that only a handful of sites had Pb levels great enough to preclude their use as gardens (Basta N, 2011, personal communication). Robust testing is, however, necessary to ensure that sites have risk levels appropriate for UA.

15.9.2 Management for Soils with Pb Contamination

A number of management practices are useful in dealing with the effects of Pb contamination in garden soils (Table 15.8). None of these practices is a complete solution to the problem, but all can provide some measure of risk abatement in UA settings. Testing soil for Pb is a must in urban gardens. Site history is a major factor in Pb levels, so testing is especially crucial in sites close to roadways and those which have been utilized previously for buildings and industrial uses. Remediation management (Table 15.8) should be applied to all UA soils with total soil Pb levels

Table 15.8 Recommended management practices (RMPs) for mitigating Pb contamination risk in urban agricultural soils

RMP	Comments
Soil test	• Necessary in all urban garden sites • EPA methods 3050B (acid digestion) and 31A (microwave digestion) are considered standard • EPA recommends <400 ppm Pb for playgrounds • Sites with >1,000 ppm Pb are not suitable for UA
Cover bare soil at garden sites	• Reduces direct ingestion and dust exposure
Amend soil pH to neutral	• Reduces bioavailability of heavy metals
Raised bed cultivation	• Reduces risk of soil ingestion and crop Pb uptake • Increased cost and resources • Subject to recontamination by surrounding environment • Continued soil testing may be necessary
Application of phosphate fertilizers	• Reduces bioavailability of Pb compounds • Potential for water pollution with water soluble forms

greater than 400 ppm, and conservatively, remediation measures may make sense in soils with total Pb levels near 200-300 ppm.

Providing physical exclusion barriers between garden soils and contaminated soils is a common strategy. In affected sites, applying a heavy mulch or cover to soil areas in pathways and uncultivated areas is an excellent way to reduce total risk from Pb (Clark et al. 2008). In cultivated areas, importing soil and OM and constructing raised beds above contaminated soils (sometimes with an underlying barrier) are often recommended and utilized strategies Hynes et al. (2001) Witzling et al. 2011). Witzling et al.'s (2011) survey of urban gardens in Chicago indicated that sites involving raised beds to deal with Pb concerns received additional benefits to their overall soil quality. A major caveat exists with raised beds as a strategy to prevent risks from soil Pb; studies have documented the recontamination of surface soils in raised beds by fine soil particles from adjacent contaminated areas (Clark et al. 2008). This trend suggests that raised beds in heavily contaminated areas should be re-tested for soil Pb every few years to ensure that Pb levels remain low.

In addition to exclusion strategies, practices that result in the chemical immobilization of Pb compounds are another tool for urban soils. Lime is commonly applied to soils with heavy metal contamination, as adjusting the pH to neutral or slightly alkaline reduces the biological availability and plant uptake of Pb and other heavy metals (Shaylor et al. 2009; Rosen 2002; Basta et al. 2001). Compost and OM are also commonly added to soils to immobilize contaminants. Composted municipal biosolids have demonstrated significant reductions in the bio-availability of Pb in both urban soils (Brown and Chaney 2003) and former smelter site soils (Basta et al. 2001). Though it is rarely mentioned in popular and extension literature on Pb contamination, the application of phosphorous (P) rich fertilizers significantly reduces the bioavailability of Pb present in the soil (Hettiarachchi and Pierzynski 2004). Rock phosphate (Basta and Gradwohl 1998), a common organic soil amendment, and phosphoric acid (Ryan et al. 2004) can significantly reduce the bioavailability of Pb compounds in highly contaminated soils. Phosphate immobilization leads to the formation of highly stable compounds (Scheckel and Ryan 2004), but caution must be taken to ensure that water soluble phosphate fertilizers are not applied in quantities that can pollute waterways (Kilgour et al. 2008).

While much of the literature on Pb remediation deals with studies that have looked at mitigating Pb at the site scale, the recent studies on airborne re-suspension of contaminated soil particles (Laidlaw and Fillipelli 2008; Fillipelli et al. 2005) suggest that in some urban areas Pb contamination of soils is so pervasive that it must be viewed as a landscape level ecological process. Small-scale remedial measures in backyards and community gardens may only provide a temporary solution, as recontamination may occur until the issue is addressed at the landscape or neighborhood level (Clark et al. 2008). Additionally, these results suggest that Pb contaminated soils should continue to be tested for recontamination after mitigation strategies have been put in place.

Urban soils are also subject to contamination from several other trace elements and numerous organic compounds. Meuser (2010) provides an excellent scientific overview of contamination issues affecting urban soils globally.

15.10 Biological Functioning in Urban Soils

The soil habitat is highly modified in urban areas, with distinct alterations in abiotic communities, plant and arthropod communities, biogeochemical cycling, and land use patterns (Byrne 2007). Given that biological properties and assays are often correlated with robust soil quality, understanding the dynamics of urban soil ecology is critical to understanding quality and function in urban soils.

Disturbances associated with urbanization and urban areas are a fundamental driver of the biological functioning of urban soils. Soil studies conducted in residential landscapes (Scharenbroch et al. 2005) and urban community gardens (Grewal et al. 2010) have documented reduced biological activity following the disturbances of landscape construction and of converting an empty lot into a garden, respectively. Following a few decades of stable conditions, sites in these surveys demonstrated increased microbial biomass C, greater potentially mineralizable N, and a stable microbial metabolic rate in older residential landscapes (>50 years) (Scharenbroch et al. 2005) and more complex nematode foodwebs in older community garden sites (Grewal et al. 2010). Surveys of soils under different urban land uses in Stuttgart, Germany observed a similar pattern where more disturbed land uses often had lower levels of soil microbial biomass C (Fig. 15.1) and N and reduced enzyme activity (Lorenz and Kandeler 2005, 2006). Decreases in biological activity following disturbance in urban soils may in some instances be associated with compaction, as indicated by higher bulk density values at more disturbed sites (Scharenbroch et al. 2005). Nonetheless, soil microbial biomass and activity have demonstrated recovery on a decadal timescale following disturbance in urban soils.

A number of studies comparing urban soils to more natural areas suggest that soil biological processes and metabolism are altered in urban soils. A study on litter decomposition in forest soils on an urban to rural gradient in New York observed that both decomposition and N mineralization rates are higher in urban areas (Pouyat et al. 1997). A similar study in North Carolina documented increased N mineralization but decreased decomposition in urban soils (Pavao-Zuckerman and Coleman 2005). A comparison of soil C and N dynamics comparing short grass prairie and urban lawns in Colorado observed greatly increased respiration in urban areas, which the authors attributed to irrigation and nutrient application at those sites (Kaye et al. 2005). These observed increases in soil metabolic rates in urban areas may be related to urban alterations in climate (Table 15.3) and in the case of decomposition rate, have also been associated with increased earthworm populations (Pouyat et al. 1997).

Urbanization and urban land uses may also affect the distribution and abundance of soil invertebrates. In the New York area, earthworm abundance can be more than 10 times greater in urban forests than in rural forest stands (Steinberg et al. 1997). Earthworms have also been greatly reduced in highly contaminated urban sites (Hartley et al. 2008). Further, total nematode biomass and predatory nematode functional groups, along with microbial biomass, have also demonstrated declines in urban soils, compared to rural areas (Pavao-Zuckerman and Coleman 2005, 2007).

15.10.1 Improving the Biological Properties of Urban Soils

Soil biological alterations observed in urban areas such as increased N mineralization and increased numbers of earthworms may be beneficial to UA, as these processes improve conditions for crop growth. In urban soils with reduced biological activity, a range of options for increasing biological activity are available.

Management strategies aimed at improving biological properties in urban soils are closely related to those mentioned for improving soil physical properties in that a key consideration is adding OM to the soil, as additions of OM stimulate microbial biomass and the soil food web. The previously mentioned management strategies of compost and biochar additions, and cover cropping increase soil biological activity and health in the soil through increasing the input of OM and C-rich biomass. Additional strategies and treatments may complement OM inputs in improving the biological condition of urban soils including: vermicompost application, microbial inoculation, and perennial plant based systems.

Biological management of soils has been central to the rise of organic urban agriculture in Cuba. Cubans have made extensive use of recycling urban green and food waste into composts and have implemented targeted microbial inoculation strategies in their efforts to improve urban soils for agriculture (Altieri et al. 1999; Treto et al. 2002). Cuban land managers have had particular success in increasing crop yields and soil health through applying bacteria based bio-fertilizers, mycorrhizal inoculation, vermicompost, and N-fixing cover crops (Treto et al. 2002). They have also initiated extensive composting efforts in their urban areas. Urban farms and organoponicos are active in composting animal wastes, plant residues and green waste, organic industrial wastes, and organic household wastes using traditional composting methods, vermicomposting and bio-digesters (Altieri et al. 1999). These amendments are then applied to soils for urban crop production. The majority of these strategies can be implemented with fairly low cost in most urban areas.

Vermicomposting is the process of creating high quality composts by feeding food wastes, green wastes and other OM to worms, primarily the tropical species *Einsenia foetida*. Vermicomposting has been implemented widely as a low cost and effective means of transforming wastes into soil amendments and an extensive body of research literature exists on the subject (see Edwards and Arancon 2004). Extensive research on the effects of vermicompost application on agricultural crops and soils has been carried out at The Ohio State University, with many positive results. Researchers at OSU have observed increased soil biological activity (Arancon et al. 2006) and improved vegetable crop growth (Arancon et al. 2005), as well as suppression of insect herbivores (Yardim et al. 2006), after applying vermicompost to crops. Vermicomposting operations can range in scale from farm-scale processing units and windrows to small bins, or worm boxes, which can be kept in small urban spaces such as kitchens, basements, or garages. *E. foetida* worms and information on how to manage them are widely available. The process is relatively simple and widely adaptable. Given the measurable benefits of vermicompost application, it appears to be an excellent technique for improving plant growth in urban soils.

15.11 Conclusion

Vacant lot soils are becoming an abundant natural resource in the Eastern and North Central U.S. Access to open land has the potential to provide social and ecological benefits in these areas (Tzoulas et al. 2007). Urban farms and community gardens can bring beauty, community engagement, improved ecosystem services, increased access to nutritious foods, and modest economic benefits to city neighborhoods. These benefits make UA an ideal land use for vacant urban lots.

There is increasing public and institutional interest in re-localizing, or regionalizing the American food system. Increased awareness of the energy costs of food production, growing concerns about the health and safety of the industrial food system, and the foreboding evidence of a changing climate and reduced energy availability in the twenty-first century have all contributed to a great surge in interest in local food systems and small scale agriculture in the U.S. Despite relative affluence, many Americans are simply not consuming enough high nutrition foods, a problem that appears to be heightened in the depressed urban areas where vacant lots are found (Wrigley 2002; CUDC 2008). These factors and the available data on the increasing number of urban gardens and farms (Table 15.1) suggest that the conditions are right for UA to become a viable form of food production in the U.S. Additionally, UA's documented successes in producing significant quantities of food (Mougeot 2005) and improving access to nutritious foods (Zezza and Tasciotti 2010) in the developing world, suggest that UA has great potential to perform well under similar conditions in the U.S.

Among the challenges of scaling UA up to impact food security in the U.S. is a general lack of data on the subject in the North American context. Researchers, educators, and producers can benefit from credible data on production potential, economic potential and food security impacts of UA in North American cities. The current information on these subjects is sparse and generally lacking in peer review. Given the unique social and biophysical conditions of the urban environment (Pickett et al. 2008; Kaye et al. 2006) there is also a need for long-term agricultural research in urban environments. Intensive production methods, management for degraded soils, low-cost management strategies, well suited crop varieties, season extension, identification of pests and pathogens, and integrated pest management (IPM) are all research areas with strong potential to benefit urban producers. Given the current increases in interest and participation in UA, it should become a key agricultural research priority in the North Central US, as well as other regions in the developed world where UA is expanding.

Despite its significance, the research information about the management of urban soils is scanty and fragmentary. While the data base has improved on conditions and processes in urban soils during the past two decades, relatively few studies have examined the effects of management-induced changes in urban soils. Given the high levels of heterogeneity found in urban soils, the best management strategies for these soils must begin with a comprehensive, site-based evaluation of their overall condition and quality (Fig. 15.3). A key factor in determining the extent of adaptive management is the previous land use that the soil has been under (Pouyat et al. 2007). Urban soils that have been under stable, ecological land uses, such as perennial vegetation, may exhibit healthy soil functioning (Grewal et al. 2010). Soils that have undergone recent heavy disturbance are likely to demonstrate degraded physical and biological properties (Scharenbroch et al. 2005). Understanding the constraints present in a given soil allows land managers to plan targeted strategies for improving those soils.

In many urban and industrial neighborhoods, the contamination of soils by Pb and other industrial compounds is likely to continue to be a major concern. In areas where contamination is a major concern, analyzing soils for heavy metals content is a priority, and a starting point for soil quality analysis and determining the suitability of parcels for agriculture usage. There is still much to be learned about contaminated urban soils. Analytical methods, biological availability, remediation methods, and larger-scale approaches to remediation are all areas that continue to need more research.

Given the small number of studies that have been conducted on the biology of urban soils and the unique biophysical conditions found in urban areas, urban soils may be a promising area for biological research. Biologically degraded soils in cities may present opportunities for continuing to develop scientific understanding

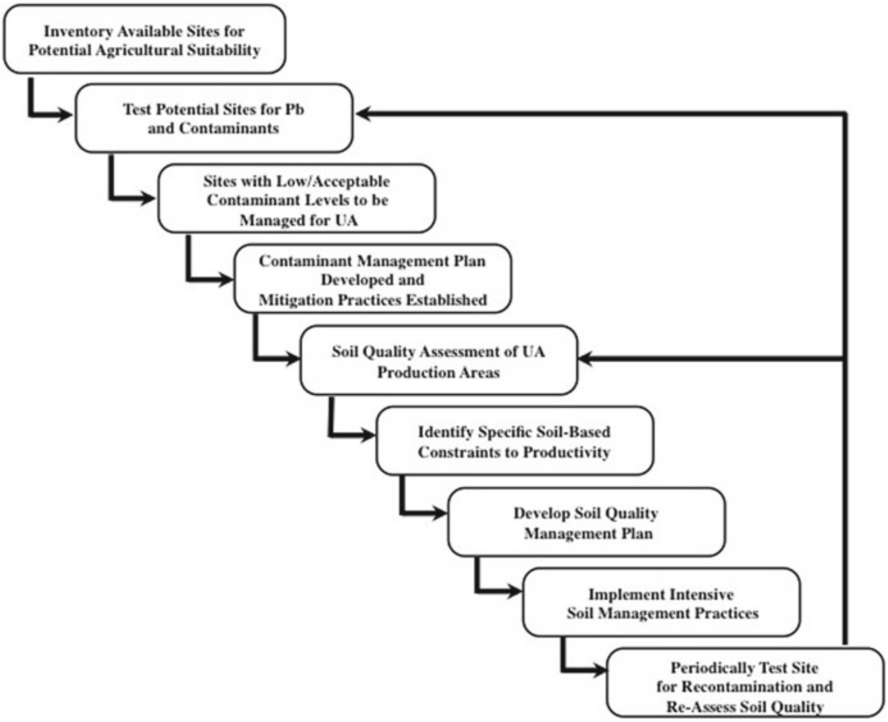

Fig. 15.3 Conceptual model for assessment and management of vacant lot soils for UA

of managing and regenerating soil biota. The recent experiences of Cuba suggest that biological management methods may be an effective means of increasing soil quality and productivity for UA (Treto et al. 2002).

A key component of managing urban soils for food production is the development of strategies and systems for capturing the large quantities of organic by-products produced in cities and transforming them into high quality soil amendments. Municipal and institutional composting programs, vermicomposting of kitchen wastes, the refinement of low cost methods of biochar production and continued education about the importance of these practices can all contribute to the knowledge to turn wastes into resources. These amendments can be a cornerstone in small scale, intensive management regimes for improving urban soils.

The current demographic trend of increasingly urban global populations suggest that urban soil management is a subject that is likely to increase in importance. Healthy soils are a necessity for ecosystem services in cities, as in all ecosystems. The unique ecological conditions in urban areas, coupled with tremendous potential for education and extension in population centers, make urban soils an exciting frontier in soil science research. Urban soils are likely to be a focus where soil scientists can extend the ecological understanding of soils as well as contribute to improved ecological and social conditions. By assessing and improving the quality of soils in cities, urban soils can be utilized as a resource to improve food security and environmental quality.

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Chapter 16
Carbon Cycle of Urban Ecosystems

Galina Churkina

Abstract Urbanization is one of the most irreversible and visible anthropogenic forces on Earth. The fraction of urban population is growing and is predicted to reach 70% of the world's population by 2050. Urban areas are both drivers and recipients of global environmental change. Carbon cycle of urban areas plays an important role in the feedbacks between urban development and global environmental change. On one side it is a driver of the global environmental change, because more than 70% of global CO_2 emissions originate in urban areas. On the other side the global and regional environmental changes such as heat waves, water scarcity, and air pollution influence urban carbon cycle. Carbon cycling through natural (e.g., urban vegetation and soils) and anthropogenic components (e.g., buildings, furniture, landfills, etc.) is intrinsically coupled in urban areas. In cities not only green plants take up carbon, but also concrete buildings. Emissions of carbon from vegetation and soils are complemented by emissions from fossil fuel burning. Urban areas have a large variety of pools to store carbon: from vegetation and soil to buildings, furniture, and landfills. The natural and anthropogenic carbon fluxes through urban areas are controlled by common drivers such as climate and urban form. Three issues are identified as the most important ones for understanding and quantification of urban carbon cycle. They include: (i) the lateral flows of carbon between an urban area and its footprint; (ii) responses of urban vegetation to urban climate and pollution, and (iii) interactions between natural and anthropogenic components of the urban carbon cycle. Because both natural and anthropogenic components are equally important for understanding urban C cycle, they have to be considered simultaneously in the design of any observation strategy or numerical model development. Understanding of the urban C cycle and the whole spectrum of its C pools and fluxes would be beneficial not only for scientists, but also for city governments. It can be instrumental in choosing the optimal policy to reduce urban C footprint.

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16.1 Introduction

16.1.1 Urban Areas as a Driver and a Recipient of Global Environmental Change

Urbanization is one of the most important, irreversible and visible anthropogenic forces on Earth. In 1950 one-third of human population lived in urban areas, in 2000 it was already one-half. Because most of the future population growth is predicted to be in the cities, by 2050 the proportion of the urban population is expected to increase to 70% worldwide (UN 2008). The interactions between human settlements and global environmental change can be grouped into two broad categories. First, human settlements are the recipients of global environmental change. Second, urban areas are the drivers of global environmental change (Fig. 16.1).

Global environmental changes such as rising air temperatures, redistribution of precipitation, increasing storm frequency, and sea level rise have multiple effects on human settlements. For instance heat waves, which are extended periods of anomalously high summertime temperatures, are especially deadly in cities because of population density and urban land surface characteristics. In summer 2003 Paris had almost 5,000 heat-related deaths, during 9 day-long heat wave (Dousset et al. 2011). Redistribution of precipitation may affect the water resources upon which urban systems rely. Future water supply in the New York metropolitan region can be adversely affected by the predicted climate change during twenty-first century (Major and Goldberg 2001).

Urban areas are also a powerful driver of global environmental change. Urbanization contributes to the changes of greenhouse gases concentrations in the atmosphere and therefore to the global warming phenomenon. As a place where people dwell and consume, urbanization is accompanied by the release of carbon dioxide, nitrous oxide, and other gases. Urban areas contribute 70% of global CO_2 emissions from energy production (IEA 2008). Urbanization modify the physical properties of the land surface (Lamptey et al. 2005; Diffenbaugh 2009), so it could cause regional to continental warming (Trusilova et al. 2008; Kalnay and Cai 2003). As an indirect effect, urban heat emissions initiate zones of enhanced atmospheric convergence and may lead to an increase in storms frequency (Bornstein and Lin 2000). Urban exhausts of particular matter influence precipitation formation processes (Kaufmann et al. 2007; Shepherd 2005; Rosenfeld et al. 2008). Pollutants of agricultural and urban origins including greenhouse gases spread over large areas and influence the biogeochemical cycles of natural and managed ecosystems sometime 100 km remote from the source (Lawrence et al. 2007; Trusilova and Churkina 2008). Emissions of NO_x from urban and agricultural land increase tropospheric ozone concentrations, which can lead to reduction of grain production (Chameides et al. 1994).

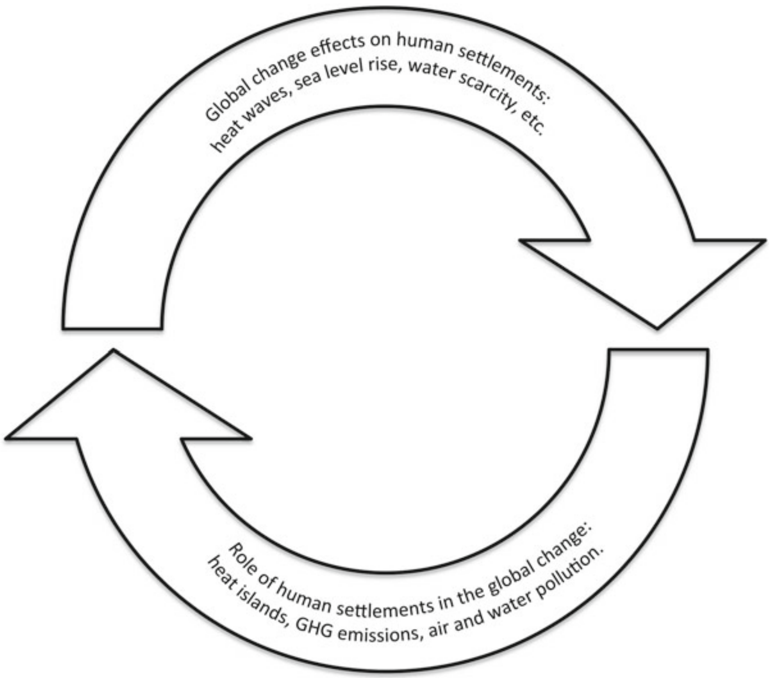

Fig. 16.1 Feedbacks between global environmental change and human settlements

Given the projections for population growth and the increasing role of cities in global environmental change, the share of population vulnerable to these changes will continue to increase. Research is needed to better understand the role of human settlements in the global environmental change and the feedbacks between the two, so that sensible mitigation and adaptation strategies can be developed.

Carbon cycle of urban areas plays an important role in these feedbacks. On one side it is a powerful driver of the global environmental change just because 70% of global CO_2 emissions originate in urban areas. On the other side it is also influenced by the global and regional environmental changes such as heat waves, water scarcity, and air pollution. Pataki et al. (2006) recently showed that the trajectory of carbon cycle of the whole North American continent will be highly influenced by patterns and forms of urban development. This chapter will describe the current state of knowledge of the urban C cycle as well as point to our knowledge deficiencies.

16.1.2 Urban C Cycle

Urban system consists of urban sprawl and urban footprint (Churkina 2008). Urban footprint is the area required to meet demands of urban population in terms of consumption and waste accumulation and the area affected by urban pollution and changes

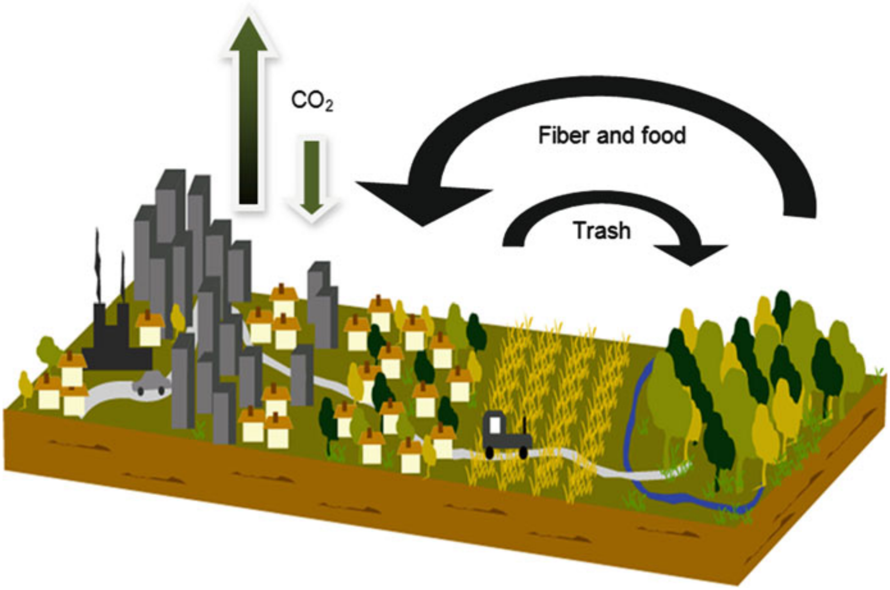

Fig. 16.2 Urban C cycle consists of carbon exchange between urban area and atmosphere as well as carbon transported between urban area and its footprint (after Churkina 2008). Green arrows refer to the natural carbon fluxes of urban areas. Black arrows refer to anthropogenic fluxes of the urban carbon cycle. Arrows in shades of black depicts fluxes from mixed (anthropogenic and natural) sources

in climate (Fig. 16.2). Unlike natural ecosystems fueled by solar energy, production of energy in urban system is closely connected with the emissions of carbon from fossil fuel burning. The amount of energy consumed per unit area per year of an urban system is 1,000 or more times greater than that of a forest (Odum 1997).

Carbon cycle of urban areas involves cycling of both organic and inorganic forms of carbon. Carbon is cycled and stored in organic form in living biomass such as trees, grasses or in artifacts derived from biomass such as wooden furniture, building structures, paper, clothes and shoes made from natural materials. Inorganic carbon or fossil carbon, meanwhile, is primarily cycled and stored in objects fabricated by people like concrete, plastic, asphalt, and bricks. The key difference between organic and inorganic forms of carbon is in how they return to the gaseous state. Organic carbon can be returned to the atmosphere through decomposition of organic matter, whereas energy input such as burning is needed to release inorganic carbon.

Cycling of carbon in urban areas based on the source of origin of each: natural or anthropogenic and is described below:

16.2 Natural Component of the Urban C Cycle

Natural component of the urban C cycle is related to the cycling of C through urban vegetation and soils. Human activities radically modify the rate at which C enters and leaves the urban ecosystems. Plants in urban ecosystems are exposed to higher temperatures (Oke 1988; Carlson and Arthur 2000) than plants in rural areas and to many pollutants. Mean annual CO_2 concentrations in a city are substantially higher (e.g. ~480 ppm in Rome in 2004 (Gratani and Varone 2005)) than comparable background concentrations (376 ppm). Emissions of NOx and VOC from urban cars and energy production lead to high tropospheric ozone concentrations in urban (Kleinman et al. 2002) and exurban (Gregg et al. 2003) areas, which can reduce plant growth. Long term studies of climate and pollutant changes along rural-urban gradient show that this gradient persist over the years (George et al. 2007) and is similar to the one predicted with global climate change. Cities themselves represent microcosms of changes that are happening globally making them useful case studies for understanding responses to global change (Grimm et al. 2008).

16.2.1 Carbon Uptake

Similar to other land or aquatic ecosystems, C enters urban ecosystems through the process of photosynthesis. The main controls over photosynthesis rate are light, atmospheric CO_2 concentration, air temperature, water availability, nitrogen (N) supply, and tropospheric ozone (O_3) concentration (Larcher 1995). Temperature governs photosynthesis reaction rates. N is required to produce photosynthetic enzymes and water is essential for general metabolism of plants. At background concentrations, O_3 is not important for plant C uptake. At certain levels, however, O_3 can damage plant leave's cells and reduce photosynthesis rate.

In general, C uptake in urban ecosystems takes place at higher CO_2 concentrations, higher temperatures, and under influence of a range of pollutants. The synergetic effect of these changes on C uptake of plant is still poorly understood. One of very few existing studies of plant productivity along a rural-urban gradient, which is characterized by gradual changes in temperatures (daytime average 3.3°C) and CO_2 (21%), shows increase of almost 115% of the productivity measured as above-ground biomass of annual plants (Ziska et al. 2004) (see Chap. 14).

16.2.2 Carbon Release

Release of C in urban ecosystems is a result of organic matter (OM) decomposition as well as plant's growth and maintenance respiration. Plant respiration provides energy for a plant to acquire nutrients and to produce and to maintain biomass. Plant respiration rate increases with increase in temperature. Maintenance respiration also depends on tissue chemistry (Reich et al. 1998). About half of C acquired by plants in the process of photosynthesis is respired back to the atmosphere (Waring and Running 1998).

Decomposition is the physical and chemical breakdown of OM such as detritus material of plants, and animals, and microbial material. It releases C to the atmosphere and nutrients into the soil for plant and microbial production. Three types of factors control decomposition: physical environment (soil temperature and moisture),

Table 16.1 Carbon residence time, uptake, and release in urban areas of the USA. Data for C sequestration and C release of urban soils, buildings, as well as C uptake of landfills and C release from urban infrastructures were not available (n/a)

	Natural component		Anthropogenic component		
	Vegetation (urban trees)	Soil	Building (wood/plastic/drywall)	Infrastructures (concrete)	Landfill
Residence time [year]	20 (1–45)[a]	1–500	12–80[b]	11.5 (0.8–100)[b]	1–500[c]
C uptake [PgC/year]	0.02[a]	n/a	n/a	0.0004[d]	n/a
C release [PgC/year]	~0.006–0.01[a]	n/a	n/a	n/a	0.09[c]

[a]Nowak and Crane (2002)
[b]Bureau of Economic Analysis, U.S. Department of Commerce (2003)
[c]Franklin Associates (1998)
[d]Gajda (2001)

the quantity and quality of substrate available to decomposers, and the characteristics of the microbial community (Chapin et al. 2002). In urban ecosystems, all these three types of factors experience influence of human disturbance. Soil temperatures in urban areas are higher because of heat island effect. Although air humidity may be lower than outside the cities (George et al. 2007), soil moisture is often relatively high, because of irrigation. The natural cycling of litter and nutrients between plants and soils is interrupted. In parks and residential yards, leaves and branches are collected and carried outside the urban ecosystems. Because litter inputs in urban ecosystems are almost negligible, decomposition rate or C release is usually lower than in natural ecosystems (Table 16.1). To compensate for low soil fertility, the vegetation is fertilized, which changes soil microbial community (see Chaps. 9 and 12).

16.2.3 Carbon Storage

Carbon is stored in urban vegetation and soils. Total amount of C stored in urban ecosystems depends on the build-up density of the city, dominant vegetation types, rates of C uptake and release by vegetation, C release by soil, as well as management of vegetation and soils. Importantly, not only soils of the open green areas are stores of C in the cities, but also soils underneath of buildings and other impervious areas can contain organic soil layers. Organic C under impervious surfaces does not decompose because of the lack of oxygen. So the centers of old towns often contain a deep layer of organic soil, which may be up to several meters deep. In contrast to these sealed soils, soils in vegetated urban areas can either accumulate or release C depending on the climate and their management regime. In a desert city, soils in urban parks and gardens tend to accumulate C, because parks are usually watered and fertilized. In contrast, soils in cities build on C rich soils in temperate latitudes are likely to lose carbon, because of lower C inputs and higher temperatures. Existing studies show a wide variation in above- and belowground C densities of urban forests and soils among cities located in different climates (Table 16.2).

Table 16.2 Estimates of above-ground and below-ground carbon density for four US cities

Variable	Unit	Seattle	Atlanta	Baltimore	Oakland
Impervious area	%	57	39.8	50.4	48
Above-ground C density	kgC/m^2	8.9	3.6	2.5	1.1
Below-ground C density	kgC/m^2	n/a	7.8	6.3	5.9

Data for Atlanta, Baltimore, and Oakland are from Pouyat et al. (2006). Data for Seattle are from Hutyra et al. (2011). Estimates of below-ground C density for Seattle were not available (n/a)

The general assumption is that the C density of urban ecosystems decreases with increasing fraction of impervious area. (Hutyra et al. 2011) confirmed this trend for aboveground live C stocks of vegetation in Seattle. The C stock density increased from 1.8 ± 1.4 kgC/m^2 for areas with high impervious surface cover to 14 ± 4 kgC/m^2 for urban forests. The live biomass of the same land cover type, e.g., coniferous forest, however did not change significantly with increasing distance from the Seattle urban core. Comparative studies of similar gradients for human settlements under different climate and socio-economic conditions are needed to determine how and if these trends change.

The potential of urban settlements to store C is underestimated. A recent study showed that human settlements of the US currently store 10% of the total C stored in all US ecosystems (Churkina et al. 2010). Noticeable differences exist between amounts of C which densely and sparsely populated urban areas can store. The average C storage in soils and vegetation in human settlements of the USA was estimated to be 0.9 PgC for urban (95,018 km^2) and 14 PgC for exurban areas (1,395,347 km^2) in 2000 (Churkina et al. 2010). One third of the urban areas of the US are impervious, e.g., paved or covered by buildings (Nowak et al. 2001). The other two thirds are green surfaces, e.g. covered by grasses or urban forests. Exurban land comprises of parcels or lots which are larger than those in urban areas, but which are generally too small to be considered as productive agricultural land use (Theobald 2005). According to a study from Michigan (Zhao et al. 2007), more than 85% of exurban surface areas are covered by green vegetation and only 5–9% are impervious. (Also see Chaps. 3, 5, and 9.)

16.3 Anthropogenic Component of the Urban C Cycle

In contrast to natural ecosystems, a substantial amount of C cycling through urban systems is directly related to human activities such as energy production, transportation, construction of buildings, and production of trash (Fig. 16.2).

16.3.1 Carbon Uptake

In urban areas not only green plants uptake C, but also concrete buildings and structures. Similar to some rocks, concrete buildings and structures absorb C in the process of carbonation. Carbonation is a chemical process, where atmospheric CO_2

is fixed as stable carbonate minerals such as calcite, dolomite, magnesite, and siderite. Atmospheric CO_2 reacts with CaO in concrete to form calcite ($CaCO_3$). This is reverse reaction of the calcination process used in cement making. The carbonation process is slow as atmospheric CO_2 has to diffuse into the solid material and dissolve in its pore fluid. The main controls behind CO_2 uptake in concrete and rocks are atmospheric CO_2 concentrations, air temperature, air humidity as well as water content, chemical composition, and porosity of materials (Gajda and Miller 2000; Kjellsen et al. 2005; Oelkers et al. 2008).

A range of studies in the USA (Gajda and Miller 2000) and Europe (Kjellsen et al. 2005) have explored and continue to explore the possibility to uptake C in concrete buildings after their demolishment. Because carbonation is a slow process, the amount of C absorbed by concrete infrastructures remains relatively small. The total amount of C which can be captured in the U.S. concrete infrastructures in 1 year is two orders of magnitudes smaller than that captured by urban forests (Table 16.1).

16.3.2 Carbon Release

Emission of CO_2 in urban areas occurs from burning fossil fuels to generate energy for heating or cooling of homes, electricity, fueling cars, etc. In the U.S., residential and commercial buildings alone account for 39% of the C emissions (Brown et al. 2008). Transportation accounts for one-third of U.S. emissions, and industry is responsible for 28%. Coal, oil, and natural gas are the three different forms of fossil fuels that are widely used. Large scale use of fossil fuels started since the middle of the nineteenth century. Presently, fossil fuels are the cheapest sources of energy available for personal and commercial uses. Petroleum is used to fuel vehicles while coal and natural gas are used to produce electricity for homes and offices. Statistics show that almost three-fourth of the demands of the energy in the world is fulfilled by fossil fuels (IEA 2010).

Not all energy consumed in urban areas is produced within the city limits. Some of it is produced at distant power plants and transported to the cities via power lines. In addition to C emitted during combustion of fossil fuels, C is also emitted during extraction, processing, and transportation. Therefore, cities are usually responsible for more CO_2 emissions than emitted immediately within its boundaries. A study of ten global cities suggests that C emissions within city boundaries are range from 45% to 95% of end-use emissions (Kennedy et al. 2009).

A range of factors controls amount of CO_2 emitted in the cities. These can be divided into two broad groups such as geophysical factors and technical factors (Kennedy et al. 2009). Geophysical factors include climate, access to resources other than fossil fuels, and gateway status. Technical factors include power generation, urban form, and waste processing. Emission of C to produce heat for urban buildings is closely related to a climate variable such as heating degrees days. Access to other resources than fossil fuels for power generation substantially reduce emissions of CO_2. For instance Geneva and Toronto have access on hydropower and relatively low intensity of emissions. Cities which have major airports or marine ports would have additional emissions from aviation and fleet. Emissions from transportation are inversely related to population density or urban form (Newman and Kenworthy 1999; Kennedy et al. 2009).

16.3.3 Carbon Storage

C is stored in infrastructures and artifacts created by people such as buildings, furniture, books, clothes, and landfills (Churkina 2008) in both organic (e.g., plant-based materials) and inorganic (e.g., concrete) forms. The amount of C stored in human infrastructures depends on its chemical composition, the length of its life-span, and the area or volume it occupies. Life-span of human artifacts ranges from years for clothes and paper products to decades for buildings (Bureau of Economic Analysis. U.S. Department of Commerce 2003), and to thousands of years for landfills. C density of anthropogenic pools in the US is high in urban (17–29 kg cm^{-2}) and low (0.7–0.4 kg cm^{-2}) in exurban areas (Churkina et al. 2010).

Multi-stores buildings in a densely populated city are mostly made of concrete and store most C in inorganic form. Use of wood in public and residential houses depends on a country's cultural traditions, wood prices, fire regulations, and climate. Residential houses in North America have most likely more organic C per unit of house area than those in Germany or France. This trend is related to the differences in wood prices, fire regulations, and associated insurance costs. Within the US, residential houses in the north have more wood per house unit area than those in the south (Wilson 2006).

Landfills can accumulate substantial amounts of C over time. Barlaz (1998) estimated the amount of carbon that remains in long-term storage after anaerobic decomposition of municipal solid waste in landfills. His estimate of the total carbon sequestration in landfills was 119 TgC/year globally. It was calculated based on waste generation rates for 1994 for the United States and the early 1990s for other countries as well as with the assumptions that waste at the landfills achieves maximum decomposition and that carbon sequestration factor does not change from country to country. Deposition of trash to landfills is associated not only with sequestration of carbon but also with the emissions of methane (CH_4), another powerful greenhouse gas. Methane emissions may diminish if not revert the advantage of long term C sequestration in landfills into disadvantage.

Cities can be designed to store more C in both organic and inorganic forms. In densely built-up cities, additional C storage in buildings would be an option. Using wood or other carbon rich construction materials for building construction and furniture would increase C storage in cities. Any option for increasing C storage in cities should be assessed together with associated CO_2 or other greenhouse gas emissions and the additional benefits or issues it may bring to urban dwellers. For instance use of wood in buildings, instead of brick, aluminum, steel and concrete, can increase C storage in human settlements and reduce emissions of greenhouse gases related to the construction and lifecycle of buildings. Production of bricks and concrete is much more energy-intensive than fabrication of wooden construction materials, and is accompanied by high CO_2 emissions from fossil fuel burning. Any increase in wood use in building construction would have implications to its production. Rising demand for wood must be accompanied by an increase in the area of forest under management for long-term sustainable timber production.

16.4 Gaps in Our Knowledge of Urban C Cycle

Although urbanization is undoubtedly one the most substantial anthropogenic forces determining continental and global C budgets, the number of empirical studies attempting to connect urbanization to global C cycle is rather limited. Most attempts to quantify effect of urban areas on C cycle have been constrained to assessments of urban CO_2 emissions (Pataki et al. 2009; Kennedy et al. 2009). While reviewing the urban C cycle, the following issues are identified as the priority ones in quantification of the role of urban areas in the global C science. These include: (i) the lateral flows of C between urban areas and their footprints, (ii) responses of urban vegetation to urban climate and pollution, and (iii) interactions between natural and anthropogenic components of the urban C cycle, and responses of vegetation to interactive effects of urban climate and pollution.

16.4.1 Lateral Flows

There are four major lateral flows of C between urban areas and their footprints. These include flows of fiber, food, fossil fuels, and trash. Flow of fiber includes construction materials, furniture, books, papers, clothes, and footware. Flow of trash includes household and industrial waste, sludge, as well as construction and demolition debris. To include these flows into the calculation of urban C, the following quantities must be known: the amount of C transported into the cities with abovementioned flows, amount of C which stays in the city and its residence time, as well how much of this C ends up in the landfills, is burned, or recycled. In addition, emissions of CO_2 associated with each life cycle stage of these products must also be quantified. Although some studies include calculations of C emissions associated with production and transportation of some of these items (Ramaswami et al. 2008), few if any have included estimates of C storage associated with each of these flows.

16.4.2 Responses of Urban Vegetation to Interactive Effects of Urban Climate and Pollution

Responses of vegetation growing in urban and suburban areas to the interactive effects of urban climate and pollution remain a substantial gap in the scientific knowledge. Given that the cities are microcosms of changes happening globally, understanding and quantifying these responses would be helpful not only for understanding urban C cycle, but also for predicting land vegetation responses to global change in general. A methodology for attributing these responses to different drivers must be established. This methodology must account for various pollutants relevant for vegetation growth (e.g., O_3, NOx, CO_2) and for climate variability of different cities. Stable- and radio-isotopes based methodology offers one of the possible tools (Wang and Pataki 2010; Hsueh et al. 2007).

16.4.3 Interactions Between Natural and Anthropogenic Components

Better understanding of interactions between natural and anthropogenic components of the urban C cycle would also be beneficial for mitigation of C emissions as well as of the urban "heat island effects".

Cities with large parks and open green space have the potential to uptake and store more CO_2 as well as to reduce the urban "heat island" effect in summer. As cities become less densely populated their ecosystem C storage is increasing relative to the anthropogenic C emissions (Fig. 16.3). Anthropogenic C emissions from transportation and residential energy use of individual cities are currently an order of magnitude higher than the net C sequestration by urban forests (Table 16.3). It implies that urban forests alone are unlikely to offset anthropogenic C emissions of cities. Its C sequestration capacity has to be considered along with other benefits of urban vegetation. For instance in addition to enhancing C storage and uptake, city trees cool surfaces by up to 10°C through shading and transpiration of water. This can reduce energy use in summer and CO_2 emissions from energy production for air conditioning. In contrast, large green spaces embedded in the cities create larger distances for people to travel to work and to shop. Unless there is a well developed and efficient public transport system, longer daily travel distances lead to higher C emissions from road transportation. Various benefits of urban vegetation have to be estimated and considered simultaneously with the costs associated with urban vegetation maintenance.

Maintenance of urban vegetation is generally associated with higher C emissions than natural vegetation due to the use of fuel-driven machinery, such as lawnmowers and petroleum-based fertilizers and higher water use. CO_2 is also emitted during fertilizer production and by the generation of energy necessary for irrigation.

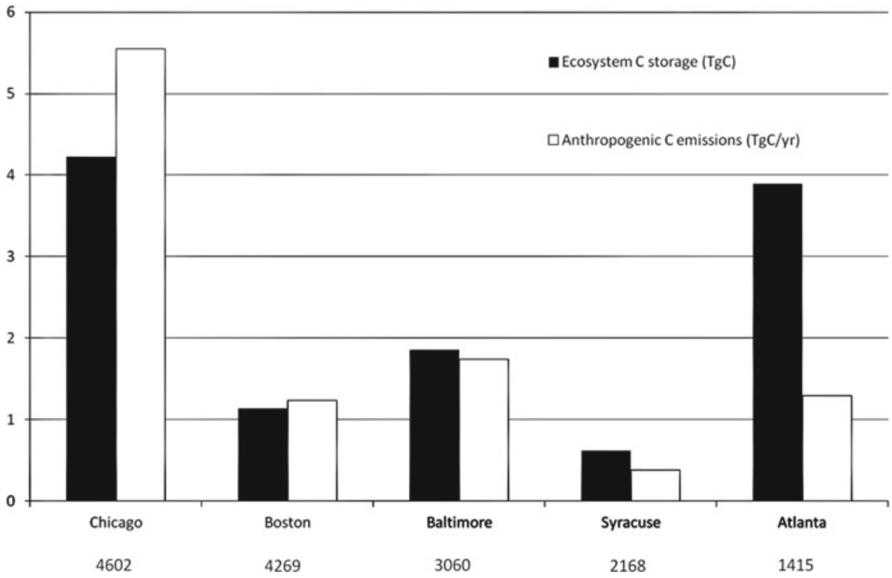

Fig. 16.3 Storage of carbon in ecosystems and anthropogenic carbon emissions of five U.S. cities. Ecosystem carbon storage is calculated as a sum of below and aboveground C storage in urban soils and vegetation using estimates from Pouyat et al. (2006). Anthropogenic C emissions are calculated as a product of per capita C emissions from transpiration and residential energy use (Brown et al. 2008) and population (U.S. Census Bureau 2010) of each city in 2005. The number under the city's name refers to its population density (persons per km^2) in 2005

In general, C will accumulate in urban gardens and parks if they are fertilized or irrigated. However, use of N fertilizer is associated with emissions of N_2O, another powerful greenhouse gas. As a result, C uptake and accumulation in a park may be accompanied by high N_2O emissions and an overall increase in greenhouse gas emissions (Townsend-Small and Czimczik 2010).

16.5 Conclusions

In contrast to natural ecosystems, both natural and anthropogenic components are equally important for understanding urban C cycle and have to be considered simultaneously. Observation strategies of urban C cycle pools and fluxes have to be designed to deal with both components simultaneously and to allow us to disentangle influences of natural and anthropogenic drivers. Numerical models of urban C cycle incorporating both components have to be developed. These models will be crucial for understanding urban C dynamics as well as for future projections of C emissions and C sequestration potential in urban areas.

Many city governments are taking active steps towards reduction and offsetting their C emissions. Their actions however mostly one-sided and only include reduction

Table 16.3 Estimates of natural C pools and fluxes and anthropogenic C emissions for five cities

			Natural C pools[b] and fluxes[c]			Anthropogenic C emissions[d]		
City	Area km²	Population in 2005[a]	C pool belowground (TgC)	C pool aboveground (TgC)	C sequestration by urban forests (TgC/year)	Transportation (TgC/year)	Residential energy use (TgC/year)	Total emissions (TgC/year)
Atlanta	341	483,198	2.671	1.22	0.032	0.789	0.507	1.296
Baltimore	209	640,064	1.323	0.527	0.011	0.867	0.869	1.736
Boston	143	609,690	0.841	0.29	0.007	0.627	0.607	1.234
Chicago	614	2,824,584	3.369	0.855	n/a	3.197	2.353	5.555
Syracuse	65	140,939	0.462	0.157	0.003	0.242	0.135	0.378

Data for C sequestration in Chicago's urban forests were not available (n/a)

[a]U.S. Census Bureau (2010)

[b]Pouyat et al. (2006)

[c]Nowak and Crane (2002)

[d]Calculated from Brown et al. (2008) and U.S. Census Bureau (2010)

of C emissions from anthropogenic sources. These actions are based on the view of urban areas as major emitter of CO_2. As we showed above cities are not only emitter but also stores of C. Therefore more complete understanding of the urban C cycle and the whole spectrum of its C pools and fluxes would be beneficial for choosing the optimal policy to reduce urban C footprint.

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Chapter 17
Legacy Effects of Highway Construction Disturbance and Vegetation Management on Carbon Dynamics in Forested Urban Verges

Tara L.E. Trammell and Margaret M. Carreiro

Abstract Natural ecosystems, like forests, adjacent to urban interstates are directly exposed to vehicle emissions (e.g., carbon dioxide) and may be strategically located for providing ecosystem services, such as carbon storage and sequestration. The primary goal of this study was to determine the effects of past soil disturbance (i.e., topsoil removal during highway construction) and an exotic shrub (Amur honeysuckle) on soil carbon dynamics in forests adjacent to urban interstates in Louisville, Kentucky. To predict potential tree and soil carbon storage and sequestration, CENTURY model simulations were conducted. A short-term (94-year) CENTURY model simulation was performed to determine the impact of past soil disturbance and honeysuckle presence on tree and soil carbon dynamics. The forest with highly disturbed soils exhibited 20% lower tree and 3% lower soil carbon storage than the forest with undisturbed soils. Tree carbon storage was 1.6 times more and soil carbon storage 1.2 times more in the forest with low honeysuckle density compared to the forest dominated by honeysuckle. Since honeysuckle removal is a management option for improving biodiversity in these interstate forests, the implication of honeysuckle eradication on long-term carbon storage in these ecosystems using 494-year CENTURY model runs was explored. Once the impact of honeysuckle was removed from the CENTURY simulations, the forest dominated by honeysuckle with intensively disturbed soils exhibited similar tree carbon storage (70,133 g Cm^{-2}) and greater soil carbon storage (7,380 g Cm^{-2}) than the forest with low honeysuckle density and undisturbed soils (tree = 71,152 g Cm^{-2}, soil = 7,053 g Cm^{-2}). These CENTURY model simulations suggest that over long time spans the removal of Amur honeysuckle could improve tree and soil carbon sequestration even where urban soils experienced intense past soil disturbance.

Keywords Urban interstates • Urban forests • Exotic invasive species • Amur honeysuckle • Lonicera maackii • Soil disturbance • CENTURY model • Carbon dynamics

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17.1 Introduction

Approximately 50% of the world's population now lives in urban environments, an increase from 29% in 1950 (United Nations Department of Economic and Social Affairs, Population Division 2009). In the United States, around 80% of the population already resides in metropolitan areas (U.S. Census Bureau 2001). While urban areas only account for 3% of United States land area (Nowak and Walton 2005), they produce 78% of the nation's greenhouse gases (Grimm et al. 2000). Cities change their local environment via the built landscape and the large inputs of matter and energy required to maintain urban ecosystems (McIntyre et al. 2000). As a result, fossil fuel combustion is high, producing elevated carbon dioxide (CO_2), nitrogen oxides (NO_x), ozone (O_3), and other air pollutants in urban atmospheres (Gatz 1991), all of which can affect primary production, and by extension carbon (C) sequestration, in urban natural areas. Since motor vehicles consume most of the fossil fuel in cities (BTS 2004), combustion product concentrations are likely to be elevated near roads, particularly urban interstates. Woody vegetation communities alongside interstates, as compared to other city locations, are directly exposed to high vehicle emission levels and thus may be "hot spots" in the urban landscape that provide greater ecosystem benefits (e.g., air pollutant capture, carbon storage and sequestration) than might be estimated by their small total area alone.

Forest communities adjacent to urban interstates not only provide ecosystem services aboveground, but belowground as well, specifically by storing C in roots and soil. In fact, C storage has been shown to be proportionally higher in forest soils than in aboveground vegetation (61% of total C storage; Birdsey and Heath 1995), especially in urban areas (78–88% of total C storage in urban greenspace, Jo and McPherson 1995). However, belowground conditions and resources that can alter C storage are also directly affected by past and current disturbance to soil. For example, roads affect the soil physical environment, and consequently C sequestration, by compaction (Jim 1998). Legacies due to soil horizon disturbance and removal during road construction also alter edaphic conditions (Trombulak and Frissell 2000). Chemical and physical conditions along highways may stress plants and soil decomposers, which will in turn affect organic matter quantity (Wellburn 1990) and quality (Findlay et al. 1996), and soil processes involved in the cycling of C and N (Pouyat et al. 1997). If trees are younger along highways than other areas (due to tree cutting when highways were built), and their growth stimulated by the higher CO_2 and N concentrations near highways (Bell and Ashenden 1997; Gilbert et al. 2003; Roorda-Knape et al. 1998), then C sequestration by roadside forests may be greater than in other wooded locations nearby. The C storage in highway soils beneath these growing trees may be amplified, potentially making forest soils, as well as the vegetation, adjacent to urban interstates important C sinks in urban environments.

Researchers have previously suggested that highway verge vegetation and soils may play potentially important roles as C sinks (Forman et al. 2003; Nowak et al. 2002) and the Federal Highway Administration has recognized the potential importance for highway rights-of-way (ROW) to sequester C by starting the Carbon Sequestration Pilot Program (CSPP) in 2008 (FHWA 2010). The purpose of this pilot project was to determine the possibility of State DOT's (Department of Transportation) to reduce C emissions and the potential to create revenue by altering vegetation management practices in the ROW. While the government has begun investigation of this potential, our study is one of the few to provide data on the ability of *roadside forests* to store and sequester C in *urban* environments. The prior research conducted by the authors on aboveground C storage and sequestration demonstrated that forests alongside urban interstates stored and sequestered more C on an area basis than previous citywide estimates across the urban landscape (Trammell 2010). These results support the importance of natural forest patches, particularly those next to interstates, in mitigating CO_2 emissions across urban landscapes.

The goal of this study was to determine current soil C storage in forests adjacent to urban interstates in Louisville, Kentucky and to estimate the extent to which soil disturbance and invasive species may affect soil carbon dynamics in these locations. Previous research conducted on the forest soils along these urban interstates identified sites with dissimilar intensity of disturbance caused by past interstate construction (Trammell et al. 2011b). For example, locations where topsoil was removed during construction are expected to have lower soil C storage than soils with natural soil horizon development. Additional research along Louisville's interstates identified an exotic invasive species, Amur honeysuckle (*Lonicera maackii*), as the most important factor in explaining forest structure (Trammell and Carreiro 2011) and ecosystem function (Trammell et al. 2011a). The data showed that aboveground biomass was reduced in plots dominated by this exotic shrub, indicating that both primary productivity and soil C content may be reduced in honeysuckle-dominated stands. Thus, forests dominated by honeysuckle were expected to have lower C storage than forests with low honeysuckle density.

To estimate C storage and sequestration the CENTURY model was used. It is a dynamic ecosystem model that simulates C and nutrient flows between plants and soils over century time scales (Parton et al. 1993). In this study, the CENTURY model was used to assess the potential of forest soils adjacent to interstates to store and sequester C, based on their current starting conditions. Predicting soil C dynamics into the future along highway verges will provide information on the continuing potential of these often highly disturbed soils to store and sequester C. The advantages of modeling soil C dynamics using CENTURY include its ability to predict whole ecosystem functioning and simulate responses to different disturbances (Parton et al. 1993). Model simulations that include human disturbance (like interstate construction) are especially valuable when attempting to model soil dynamics in urban environments. Prior to conducting predictive CENTURY model simulations (tree and soil carbon storage and sequestration), the model runs focused on calibrating CENTURY, previously developed for natural systems (e.g., grasslands, forests), to the highly modified soils in forests adjacent to urban interstates. After calibrating CENTURY to the forest plots adjacent to urban interstates, model simulations were used to predict tree and soil C storage and sequestration in three forest plots along Louisville interstates. The three forest plots were chosen by their opposing vegetation

Photo 17.1 Photo exhibits a high-density honeysuckle plot with undisturbed soil adjacent to I-64 in Louisville, KY (Photo courtesy T. Trammell)

structure (low-density versus high-density honeysuckle) and past soil disturbance (sub-soil versus undisturbed soil; see Sect. 17.2.2). The three forests plots were classified as high-density honeysuckle plot with sub-soil (HD-SS), high-density honeysuckle plot with undisturbed soil (HD-U) (Photo 17.1), and low-density honeysuckle plot with undisturbed soil (LD-U).

CENTURY simulations were conducted at on short-term (94-years) and long-term (494-years) time scales. The short-term model simulations determined the differences in C dynamics between plots with highly disturbed soils (i.e., topsoil removal) and plots with natural soil horizon development, and between plots dominated by honeysuckle and plots with low honeysuckle density. The long-term model simulations were used to investigate the length of time needed for plots with lower C storage and sequestration to achieve more equivalent C storage to plots with greater initial carbon storage. Since the removal of honeysuckle in forests along urban interstates is an option for improving ecosystem structure and function (Trammell 2010), additional long-term simulations were performed to test the impact of honeysuckle removal on C dynamics in these forests. This modeling study demonstrates how the CENTURY model responds to past soil disturbance and honeysuckle invasion in predicting C dynamics in forests along urban interstates and indicates the importance of these factors (i.e., soil disturbance and invasive species) on C storage and sequestration in these forests (Photo 17.2).

Photo 17.2 Photo exhibits a dense thicket of Amur honeysuckle adjacent to a road in Louisville, KY. This photo shows honeysuckle blooming in early spring (March 2009) before other species (Photo courtesy M. Carreiro)

17.2 Methods

17.2.1 Study Area

Louisville (38° 15′ N, 85° 45′ W) is located along the Ohio River in the Interior Low Plateau, Bluegrass Section and the Eastern Broadleaf Forest biome (The National Atlas of the United States 2009). Louisville has a total population of 713,877 with a mean density of 695 persons km^{-2} (U.S. Census Bureau 2008). The mean annual precipitation is 113 cm, which is evenly distributed throughout the year. The mean annual temperature is 13.8°C (56.9°F), the mean minimum temperature in January is −3.9°C (24.9°F), and the mean maximum temperature in July is 30.6°C (87.0°F; National Climatic Data Center 2009).

17.2.2 Louisville Interstate Plots

The Louisville metropolitan area has three interstate highways that extend east (I-64), south (I-65), and northeast (I-71) from the city center. Previous research on woody vegetation composition and soil characteristics was conducted on 21, 100-m^2 plots adjacent to I-64, I-65, and I-71. Plots were selected in a stratified random manner within 1 km intervals from the city center to the county boundary for each highway. Detailed descriptions of the three interstates and the vegetation and soil data from the forest study plots are described in Trammell and Carreiro (2011) and Trammell et al. (2011b).

Trammell and Carreiro (2011) reported that the exotic, invasive shrub species, Amur honeysuckle, was the primary factor explaining variation in vegetation composition among these forested highway verges. Thus, a working definition of honeysuckle dominance was developed to investigate differences in C storage and sequestration between plots in two categories, those dominated and those not dominated by Amur honeysuckle. High-density honeysuckle plots (HD) were defined by the total honeysuckle stem density (>80 stems plot^{-1}) and the honeysuckle importance value (IV > 90; where IV = [((relative stem density) + (relative stem density > 2 m height)) * (100/2)]), and all remaining plots were defined as low-density honeysuckle plots (LD).

Pedon descriptions made by the USDA Natural Resources Conservation Service (NRCS) for each plot included information on soil classification and horizon characteristics and were used as the basis for determining soil disturbance type (Trammell et al. 2011b). Forest plots described in this CENTURY model study represented two of the four disturbance categories: sub-soil (SS) and undisturbed (U). Interstate construction practices in some locations resulted in sub-soils (most likely C-horizon material) becoming exposed at the surface, because previous A- and B- horizons were removed during construction. Undisturbed soils are those that show normal soil horizon development, and appear to not have experienced intense physical disturbance during interstate construction.

17.2.3 CENTURY Model Description

The CENTURY model is a process-based biogeochemistry model, simulating C and nutrient (N, P, and S) dynamics over long time spans (100–10,000 years) in grasslands, croplands, savannas, and forests (Parton et al. 1987, 1993). CENTURY model runs predict soil organic matter, plant productivity, and nutrient cycling using climatic variables, site soil conditions, and ecosystem-level plant data. The C flows are divided into soil and plant residue compartments, where the soil organic matter (SOM) is divided into three main compartments: active, slow, and passive. The primary driver controlling decomposition and stabilization of the soil C pools is soil texture. Average monthly temperature and precipitation also control decomposition rates. Plant C pools are divided into surface (shoot) and soil (root) litter pools, which are both further subdivided into structural (resistant to decomposition) and metabolic (readily decomposable) plant residues. Forest aboveground dynamics are represented by a single forest type and are separated into five compartments: leaves, fine branches, large wood, fine roots, and coarse roots with C and nutrients allocated to each plant part. The N dynamics follow the same flow path as C and it is assumed that most of the N is bonded to carbon (i.e., most of the total soil N pool is organic).

17.2.4 CENTURY Data Input and Calibration

CENTURY was developed to run with minimal site-specific factors and most parameters in the model are fixed and meant to remain constant in the majority of applications. The minimum input needed to run the model include site latitude and longitude, soil texture, plant lignin content, plant N content, and climate data (monthly mean precipitation and mean maximum and minimum air temperature; Metherell et al. 1993). The remaining site-specific parameters are related to plant physiological functions that control growth, death, turnover, and nitrogen inputs. The primary factors that drive differences between sites in model runs are soil texture and average monthly temperature and precipitation. The CENTURY model runs conducted on forest plots adjacent to urban interstates used average monthly precipitation and temperature data for Louisville (30 year means from the National Climatic Data Center 2009) and soil data (e.g., pH, soil texture, and bulk density) collected in each forest plot (see description below for details). The plant physiological data used were default values provided for temperate deciduous forests (since most of these parameters have not been previously measured in urban forests, e.g., C allocation to roots).

To calibrate the CENTURY model output to forest interstate plots, data collected in each plot during the summer of 2006 and 2007 were used. Site-specific tree data were collected according to the Urban Forestry Effects (UFORE) model protocol (see Sect. 17.2.4.1; Nowak et al. 2001) and used the UFORE estimated tree C storage to calibrate tree C storage in CENTURY. Soil C data collected in each plot (see Sect. 17.2.4.2) was used to calibrate CENTURY soil C storage.

17.2.4.1 UFORE Data Collection

The Urban Forestry Effects (UFORE) model was developed by researchers at the USDA Forest Service to provide a tool for managers and researchers to estimate urban forest structure and function (Nowak and Crane 2000). The UFORE model estimates tree C storage and sequestration based on forest structure (above- and below-ground living tree biomass), thus providing tree C estimates for calibrating CENTURY (Nowak et al. 2002). Urban forest field data were collected according to Nowak et al. (2001). In each plot, size structure and species identifications were determined for all trees with a diameter at breast height (dbh) greater than or equal to 2.54 cm. UFORE calculated the size structure of the tree community using tree dbh (height at 1.37 m), total tree height and height to live crown base using a clinometer, and canopy crown width (two measurements along perpendicular axes). Tree canopy condition was determined by estimating the percent of the canopy missing (% of crown volume missing leaves, estimated to the nearest 5%) and the percent dieback in the live crown area (excluding natural dieback due to self-shading).

17.2.4.2 Soil Data Collection and Sample Analyses

Soil samples were collected from June to October 2006. Each 10×10 m plot was divided into sixteen 2.5×2.5-m quadrats or eight 2.5×5-m belt quadrats, depending on the variables measured (Trammell et al. 2011b). The 5-m side of the belt quadrats were oriented parallel to the road edge (a bisection of the 10 m side of each plot). In each of the sixteen 2.5×2.5-m sub-quadrats, soil samples were collected for pH and texture analyses with a 2-cm diameter corer to a 15-cm depth. Each soil sample consisted of six pooled cores, three cores in each of two random locations per quadrat. Soil pH was measured using a 1:1 soil (15 g) to water (15 ml) mixture (pH: Orion meter #420, Beverly, MA) after mixing for 30 min. Soil texture was measured using a modified hydrometer method (Elliott et al. 1999; Gee and Bauder 1986) to determine % sand, % silt, and % clay. Bulk density measurements were obtained using an intact soil corer. Eight soil cores, 5.8 cm diameter×10–15 cm long, were collected in random locations within each 2.5×5-m belt quadrat. For bulk density (g cm^{-3}) measurements, soil length in each core was measured prior to oven drying at 105°C for at least 24 h.

A pooled site–level soil sample (proportionally weight-based from the 16 sub-quadrats) was used to determine % total C and % carbonate-C. Soils were first sieved using a 2 mm stainless-steel sieve, and subsamples were ground in a Wiley Mill to pass through a #20 mesh. Subsamples were analyzed for total C concentration using a Perkin-Elmer 2400 Series II CHNS/O analyzer (Shelton, CT, USA). Appropriate corrections were made for some sites which could have high limestone ($CaCO_3$) content to remove the inorganic C concentration from the total % C measurements, since it could overestimate the organic C content compared to locations where limestone was not the dominant parent material. Thus, soil carbonate was analyzed in each soil sample. Soil carbonate was measured using the calcium carbonate equivalent method (Loeppert and Suarez 1996) by the University of Kentucky, College of Agriculture, Regulatory Services; Lexington, KY, USA. The % organic C fraction was then calculated by subtracting the inorganic carbonate-C fraction from the total C fraction in each sample.

17.2.5 CENTURY Calibration and Simulations

To establish all starting tree and soil C pools in CENTURY, a simulation was conducted that runs over approximately 1,900 years (equilibrium run) allowing the C pools to reach an equilibrium state under natural conditions. Since tree production and soil C accumulation would be overestimated by CENTURY without positing disturbance events during this equilibrium run, natural disturbances (e.g., windstorm, fire) were simulated during the period 0 C.E. to 1900 C.E. to achieve C levels that matched those of the current forest plots. A natural disturbance (i.e., cool surface fire) was simulated every 400 years and a light windstorm in 1900 during the equilibrium run in order to reach natural conditions for the forest verges (Table 17.1).

Table 17.1 CENTURY events for equilibrium model run, calibration model runs, and model simulations in forest plots adjacent to I-64

	Time period	Natural and highway disturbance events
Equilibrium	0–1900	Forest growth with surface fire every 400 years
	1900	Light windstorm to reflect natural disturbance
	1900–1957	Forest growth
Calibration	1957	Soil erosion event and aboveground reduction in plant biomass to reflect large disturbance due to initial interstate construction
	1958–1965	Forest growth with soil erosion to reflect soil disturbance during highway construction period
	1966–1975	Forest growth
	1974	Windstorm reflecting 1974 tornado in Louisville, KY
	1975–2005	Forest growth
Simulation	2006–2100	Forest growth with reduction in maximum production rates per plot proportionate to observed production measured as biomass and LAI in forest plots
Simulation	2006–2500	Long-term forest growth simulations and long-term forest growth simulations without new maximum production rate parameter

Note: Equilibrium model run with soil series parameters
Calibration and Simulation model runs are conducted with plot specific soil parameters

Table 17.2 Soil texture (% sand, silt, and clay), bulk density (g cm^{-3}), and litterfall biomass (g m^{-2}) and leaf-area index (m^2 leaf area m^{-2} ground area) collected in autumn litterfall (2006) for each forest plot modeled by CENTURY

Plot type	% sand	% silt	% clay	% silt + % clay	Bulk density (g cm^{-3})	Litterfall (g m^{-2})	Litterfall LAI (m^2 m^{-2})
HD-SS	8.3	61.6	30.1	91.7	1.325	320.18	2.93
HD-U	11.3	72.8	15.8	88.6	0.983	NA	NA
LD-U	10.2	65.2	24.6	89.8	1.083	441.97	5.11

Litterfall was not collected in plot HD-U, thus mean total biomass and LAI for high-density honeysuckle plots was applied to these two plots to determine their reduction in maximum production rate for CENTURY model simulations

HD-SS High-density honeysuckle, disturbed soil (sub-soil), *HD-U* High-density honeysuckle, undisturbed soil, *LD-U* Low-density honeysuckle, undisturbed soil

As a first attempt to model tree (above- and below-ground living tree C) and soil (soil organic matter in total) C storage and sequestration in forests adjacent to urban interstates in Louisville, KY, CENTURY model runs were conducted on data acquired from a subset of forest plots adjacent to I-64. The three forest plots were chosen based on their having either of two related soil series (i.e., Crider or Caneyville), which have similar soil properties (e.g., soil texture, acidity, drainage). To test differences in the potential impacts of the honeysuckle shrub and soil disturbance on C storage and sequestration, the three forest plots were also chosen by their dissimilar vegetation structure (low-density versus high-density honeysuckle) and past soil disturbance (sub-soil versus undisturbed soil; Table 17.2). The three forests plots

Table 17.3 Total tree carbon storage (g m^{-2}) estimated by UFORE and total soil carbon storage (g m^{-2}) measured in each plot compared to CENTURY estimated tree and soil carbon storage in the year 2006 (field data collection)

Plot type	UFORE tree C (g m^{-2})	Measured soil C (g m^{-2})	CENTURY tree C (g m^{-2})	CENTURY soil C (g m^{-2})	Tree C difference (g m^{-2})	Soil C difference (g m^{-2})
HD-SS	4,550	3,989	5,558	3,894	+1,008	−95
HD-U	10,525	4,564	11,026	4,546	+501	−18
LD-U	37,456	4,584	36,793	4,656	−663	+72

Tree carbon (C) difference: CENTURY estimated – UFORE estimated tree C ($p<0.05$); Soil carbon difference: CENTURY estimated – measured soil C ($p<0.05$)

HD-SS High-density honeysuckle, disturbed soil (sub-soil), *HD-U* High-density honeysuckle, undisturbed soil, *LD-U* Low-density honeysuckle, undisturbed soil

chosen for the CENTURY modeling were classified as high-density honeysuckle plot with sub-soil (HD-SS), high-density honeysuckle plot with undisturbed soil (HD-U), and low-density honeysuckle plot with undisturbed soil (LD-U).

Prior to conducting predictive simulation model runs into the future, CENTURY model runs were performed for the interval between 1957 (when highway construction started) and 2006 using measurements of soil characteristics that were obtained for each forest plot (i.e., soil texture, bulk density, and pH). The construction of I-64 was initiated during the late 1950s (KYTC Projects Archive 2008), and so a disturbance event associated with interstate construction in 1957 was simulated (i.e., soil erosion and loss of aboveground plant biomass). Pedon descriptions by the NRCS showed that removal of the A- and B-horizon was one of the disturbance categories caused by interstate construction. One of the forest plots (i.e., HD-SS) used for CENTURY modeling had this type of disturbance with the topsoil removed during construction. Thus, a larger highway construction disturbance was simulated for this forest plot than for the other two forest plots. Following the initial major disturbance caused by highway construction, an 8-year period of soil erosion was assumed due to continued disturbance during interstate construction. Forest growth was simulated to occur from 1966 until 2006 with one additional natural disturbance known to have occurred at two of these plots (HD-SS and HD-U), namely a tornado in 1974. To more accurately model C sequestration and storage dynamics in these forests using CENTURY, CENTURY model output was calibrated with measured soil C data and tree C data estimated by UFORE. Slight manipulations in the disturbances (i.e., highway construction, 1974 storm) were implemented to calibrate CENTURY soil and tree C output to the measured soil C data and the estimated UFORE tree C storage data. After employing these disturbance manipulations, the CENTURY-estimated soil C content were within 95 gm^{-2} of soil C measured at the plots, and the CENTURY-estimated tree C values were within 1,008 gm^{-2} of the UFORE estimated tree C values (Table 17.3).

To predict potential tree and soil C sequestration, CENTURY forest growth simulations were conducted from the year 2006 to the year 2100. Previous research in these forests showed that primary production was reduced in high-density honeysuckle plots compared to low-density honeysuckle plots (Trammell et al. 2011a). To test the impact of this lower productivity on tree and soil C sequestration, the observed decrease in productivity was applied to a parameter in CENTURY that controls production rates (i.e., maximum production rate (MPR), hereafter referred to as the MPR parameter). The average percent reduction in measured litterfall biomass and leaf-area index (LAI) between the high-density and the low-density honeysuckle plots was applied to the MPR parameter of the low-density honeysuckle plot to determine the new MPR parameter for high-density honeysuckle plots. When individual plot measurements were not available (HD-U not used in the litterfall production study), the average high-density and low-density honeysuckle biomass and LAI data for all plots were used to calculate the reduction in MPR parameter.

The CENTURY model simulations were also conducted on the three forest plots (HD-SS, HD-U, and LD-U) to the year 2500 to determine whether plots that demonstrated lower tree and soil C storage and sequestration could eventually sequester and store similar amounts of C to the forest plot exhibiting the highest C storage. All parameters remained the same as the previous forest growth simulations to the year 2100. To test whether the potential *removal of honeysuckle* from high-density honeysuckle plots could alter C storage and sequestration in these long-term model simulations, the effect of the honeysuckle reduction on aboveground production was removed (new MPR parameter removed).

17.3 Results

17.3.1 Soil Carbon

Across all forest plots along I-64 (n = 10), inorganic and organic soil C data differed between soil disturbance categories and between low-density versus high-density honeysuckle plots. As expected, soils with undisturbed soil horizon development had greater organic (1.16 kg C m^{-2}) and inorganic (0.09 kg C m^{-2}) soil C contents than soils that were highly disturbed during interstate construction (sub-soil exposed at the surface; Fig. 17.1). High-density honeysuckle plots had lower organic (0.78 kg C m^{-2}) and inorganic (0.17 kg C m^{-2}) soil C than low-density honeysuckle plots (Fig. 17.2). Please note that statistical analyses were not performed due to low or unequal sample size. Data are shown to demonstrate initial soil C differences between soil disturbance categories and honeysuckle dominance level (high-density versus low-density) in all forest plots along I-64 (n = 10), prior to performing CENTURY model simulations.

17.3.2 Initial Carbon Storage (2006)

The initial tree and soil C storage for the CENTURY forest simulations (2006–2100) differed among the three forest plots. The highest tree (37,456 g m^{-2}) and soil (4,584 g m^{-2}) C storage was observed in the low-density honeysuckle plot

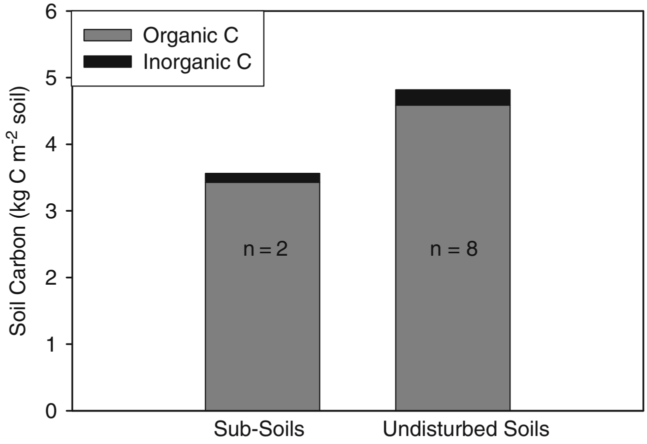

Fig. 17.1 Inorganic and organic soil C (kg C m^{-2} soil) in upper 15 cm of soils along I-64 in Louisville, KY (n = 10). Soil carbon is shown for disturbed soils (sub-soils) and undisturbed soils. Note the unequal sample size between soil disturbance categories

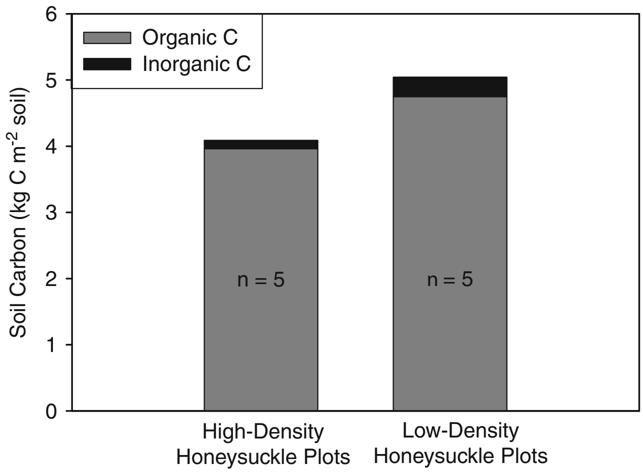

Fig. 17.2 Inorganic and organic soil C (kg C m^{-2} soil) in upper 15 cm of soils along I-64 in Louisville, KY (n = 10). Soil carbon is shown for high-density and low-density honeysuckle plots

with undisturbed soil (LD-U; Table 17.3). The high-density honeysuckle plot on exposed subsoil (HD-SS) exhibited the lowest tree (4,550 g m^{-2}) and soil (3,989 g m^{-2}) C storage (Table 17.3). The high-density honeysuckle plot with the undisturbed soil horizon (HD-U) had intermediate tree and soil C storage (Table 17.3).

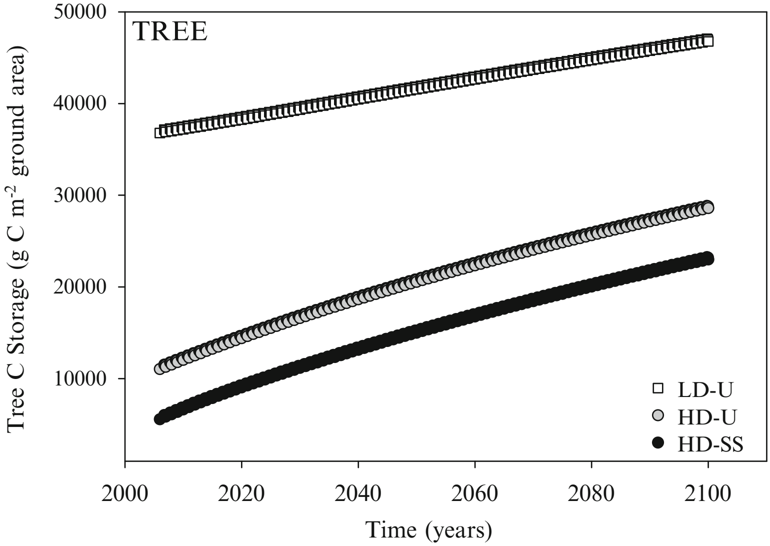

Fig. 17.3 CENTURY estimated tree carbon storage (g C m^{-2} ground area) for forest plots alongside I-64 (20 cm depth). The low-density honeysuckle plot with undisturbed soils (LD-U) is shown in *white*, the high-density honeysuckle plot with undisturbed soils (HD-U) is shown in *gray*, and the high-density honeysuckle plot with sub-soils (HD-SS) is shown in *black*

17.3.3 CENTURY Simulations (2006–2100)

The CENTURY forest growth simulations were conducted with the reduced MPR parameter in high-density honeysuckle plots. These forest growth simulations resulted in higher tree C storage (46,770 g C m^{-2}) occurring in the low-density honeysuckle plot on undisturbed soil (LD-U) than the high-density honeysuckle plot on undisturbed soil (HD-U; 28,583 g C m^{-2}) after 94 years (Fig. 17.3). However, the total amount of C sequestered was greater in the high-density honeysuckle plot with undisturbed soil (17,557 g C m^{-2}) than the LD-U plot (9,977 g C m^{-2}). In contrast, the LD-U sequestered more soil C after 94 years (731 g C m^{-2}) than the HD-U plot (139 g C m^{-2}; Fig. 17.4). After 94 years of forest growth, the LD-U plot had higher soil C storage (5,389 g C m^{-2}) compared to the HD-U plot (4,684 g C m^{-2}).

The CENTURY simulations predicted lower tree C storage in the high-density honeysuckle plot with disturbed soils (HD-SS; 22,992 g C m^{-2}) compared to the HD-U plot (28,583 g C m^{-2}; Fig. 17.3) after 94 years of forest growth. However, the total amount of tree C sequestered was similar between the HD-SS plot (17,434 g C m^{-2}) and the HD-U plot (17,557 g C m^{-2}). In contrast, soil C storage was similar between the HD-SS (4,545 g C m^{-2}) and the HD-U plot (4,687 g C m^{-2}) after 94 years of forest growth, whereas the total amount of soil C sequestered was greater in the HD-SS plot (646 g C m^{-2}) than the HD-U plot (139 g C m^{-2}; Fig. 17.4).

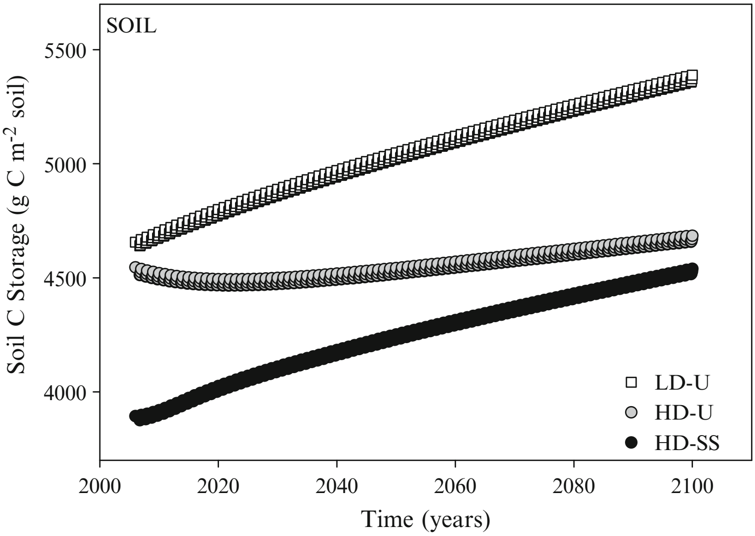

Fig. 17.4 CENTURY estimated belowground carbon storage (g C m^{-2} soil) for forest plots alongside I-64 (20 cm depth). The low-density honeysuckle plot with undisturbed soils (LD-U) is shown in *white*, the high-density honeysuckle plot with undisturbed soils (HD-U) is shown in *gray*, and the high-density honeysuckle plot with sub-soils (HD-SS) is shown in *black*

17.3.4 CENTURY Simulations (2006–2500)

To determine whether the high-density honeysuckle plot with sub-soils could store and sequester equivalent tree and soil C to the high-density honeysuckle plot with undisturbed soils, CENTURY simulations were performed until the year 2500 with all model parameters remaining the same as in the 94 year model runs (e.g., MPR adjustments in place). After 494 years of forest growth, tree C storage was lower in the HD-SS plot (52,288 gC m^{-2}) than the HD-U plot (54,535 gC m^{-2}), whereas the soil C storage was greater in the HD-SS plot (5,979 gC m^{-2}) than the HD-U plot (5,543 gC m^{-2}). The total amount of tree C sequestered was similar between the HD-SS plot (46,729 gC m^{-2}) and the HD-U plot (43,509 gC m^{-2}), whereas the total amount of soil C sequestered was greater in the HD-SS plot (2,086 gC m^{-2}) than the HD-U plot (998 gC m^{-2}).

These CENTURY simulations also demonstrate whether the high-density honeysuckle plot could store and sequester equivalent tree and soil C to the low-density honeysuckle plot. The low-density honeysuckle plot with undisturbed soils (LD-U) had greater tree C storage (71,152 gC m^{-2}) compared to the high-density honeysuckle plot with undisturbed soils (HD-U; 54,535 gC m^{-2}) after 494 years of forest growth (Fig. 17.5). However, it was observed that the LD-U plot with the greatest

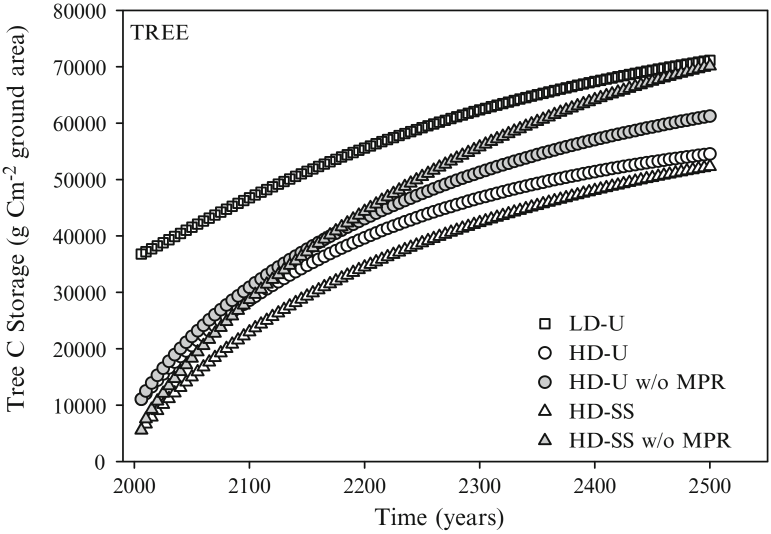

Fig. 17.5 CENTURY predicted tree carbon storage (g C m^{-2} ground area) for forest plots alongside I-64 (20 cm depth) to the year 2500. The low-density honeysuckle plot with undisturbed soils (LD-U) is shown in *white squares*, the high-density honeysuckle plot with undisturbed soils (HD-U) is shown in *white circles*, and the high-density honeysuckle plot with sub-soils (HD-SS) is shown in *white triangles*. The simulations without the adjusted MPR parameter are shown in *gray* for the HD-U plot (HD-U w/o MPR) and the HD-SS plot (HD-SS w/o MPR)

initial tree C storage exhibited lower total tree C sequestered (34,358 g C m^{-2}) than the HD-U plot (43,509 g C m^{-2}). The LD-U plot stored more soil C (7,053 g C m^{-2}) than the HD-U plot (5,543 g C m^{-2}), and the total amount of soil C sequestered was greater in the LD-U plot (2,397 g C m^{-2}) than in the HD-U plot (998 g C m^{-2}; Fig. 17.6).

To examine whether the *removal of honeysuckle* from high-density honeysuckle plots could alter C storage and sequestration in these long-term model simulations, the effect of the honeysuckle reduction on aboveground was removed (new MPR parameter removed). Once the removal of honeysuckle was incorporated into the CENTURY simulations, the total amount of tree C sequestered was greater in the HD-U plot (6,710 g C m^{-2}) and the HD-SS plot (17,845 g C m^{-2}) than in the previous model simulations, which resulted in greater tree C storage (e.g., HD-U versus HD-U w/o MPR; Fig. 17.5). Similarly, the total amount of soil C sequestered was greater in the HD-U plot (428 g C m^{-2}) and the HD-SS plot (1,400 g C m^{-2}) resulting in greater soil C storage once the impact of honeysuckle was removed from the simulations (Fig. 17.6). These model simulations also demonstrated that the HD-SS plot exhibited similar tree C storage (70,133 g C m^{-2}) to the low-density honeysuckle plot with undisturbed soils (71,152 g C m^{-2}; Fig. 17.5). The HD-SS

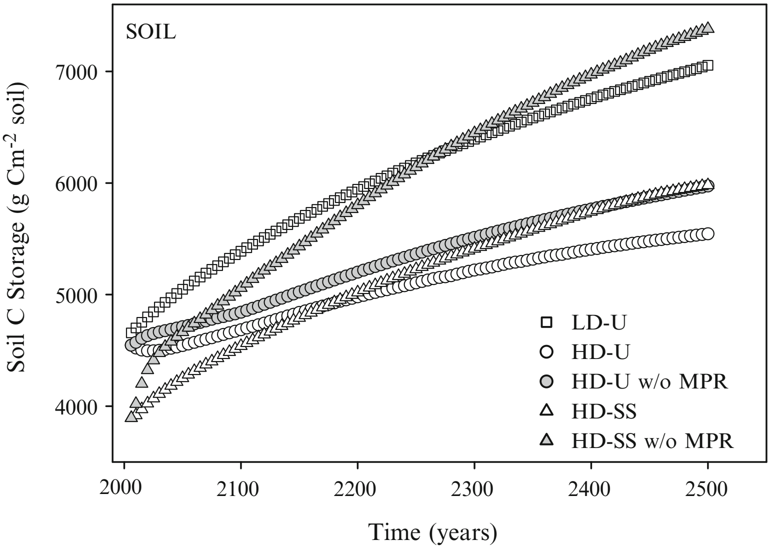

Fig. 17.6 CENTURY predicted soil carbon storage (g C m^{-2} soil) for forest plots alongside I-64 (20 cm depth) to the year 2500. The low-density honeysuckle plot with undisturbed soils (LD-U) is shown in *white squares*, the high-density honeysuckle plot with undisturbed soils (HD-U) is shown in *white circles*, and the high-density honeysuckle plot with sub-soils (HD-SS) is shown in *white triangles*. The simulations without the adjusted MPR parameter are shown in *gray* for the HD-U plot (HD-U w/o MPR) and the HD-SS plot (HD-SS w/o MPR)

plot resulted in greater soil C storage (327 g C m^{-2}) than the LD-U plot, surpassing the LD-U plot after 264 years (Fig. 17.6).

17.4 Discussion

It is suggested that highway verge vegetation and soils may play potentially important roles as C sinks (Forman et al. 2003; Nowak et al. 2002). This project is the first study to estimate the ability of roadside forests to provide this ecosystem benefit to society. CENTURY modeling of forests alongside urban interstates demonstrated the potential of these systems to store and sequester large amounts of C compared to other urban forests. Another study that used CENTURY to model urban forest C dynamics of Douglas-fir trees found carbon storage values (24,243 g C m^{-2}; Ames and Lavkulich 1999) that are 34% lower than the C storage estimates predicted for forests adjacent to Louisville interstates (initial tree C storage in LD-U = 36,793 g C m^{-2}). This supports the importance of forests adjacent to urban interstates in mitigating CO_2 emissions across urban landscapes. However, this study showed that an exotic invasive shrub, Amur honeysuckle, could reduce the ability of forests to provide ecosystem services, such as C storage and sequestration.

17.4.1 Initial Soil Carbon

The distribution of forests located alongside urban interstates in Louisville, KY was not conducive for conducting a study with an even allocation of soil disturbance categories and with sufficient quantity of each honeysuckle density plot type in each soil disturbance category (Trammell 2010). Therefore, statistical analysis of initial soil C data could not be performed (Figs. 17.1 and 17.2). However, the soil C measurements in each forest plot does permit understanding of the differences that existed between the soil disturbance categories and between high-density and low-density honeysuckle plots prior to conducting CENTURY model runs. As expected, sub-soil plots had lower soil C initially (Fig. 17.1), since a substantial amount of the topsoil was removed during the interstate construction. Similarly, the high-density honeysuckle plots had lower initial soil C than low-density honeysuckle plots (Fig. 17.2). This was also expected, since lower aboveground biomass was measured in forests dominated by honeysuckle (Trammell et al. 2011a), suggesting lower C inputs to the soil. These initial soil C differences indicate the importance of including soil disturbance categories and invasive species presence in the design of future research conducted on forests adjacent to urban interstates.

17.4.2 CENTURY Simulations (2006–2100)

CENTURY model simulations conducted on forests adjacent to urban interstates demonstrated the potential effect of past soil disturbance and the effect of a dominant exotic invasive species on tree and soil C storage and sequestration. Once CENTURY was calibrated to data measured in 2006, carbon dynamics were simulated in three forests adjacent to Louisville interstates to the year 2100 including the detrimental effect of honeysuckle on aboveground production. As expected, the presence of honeysuckle restrained forest C sequestration, such that the high-density honeysuckle plot (HD-U) did *not* establish similar tree and soil C storage to the low-density honeysuckle plot (LD-U), at least within the 94-year simulation period. However, the tree C *sequestration* was greater in the high-density honeysuckle plot than the low-density honeysuckle plot (Fig. 17.3). This is a result of the CENTURY model structure. Forest production in the CENTURY model is based on several environmental factors (i.e., moisture, soil temperature) and live leaf-area-index (LAI), and greater LAI values result in a relatively lower effect on plant production (Metherell et al. 1993). Thus, forests adjacent to urban interstates with greater initial tree biomass and C storage (i.e., low-density honeysuckle plot) will *accumulate* relatively less C in CENTURY model runs compared to forests with lower initial tree biomass and C storage (i.e., high-density honeysuckle plots).

While the sequestration of tree C was related to the initial carbon storage, soil carbon sequestration did not depend on the amount of C stored in soil pools. Soil texture (% clay) was the primary predictor of the sequestration of soil C in these forests adjacent to urban interstates (Trammell 2010), which is not surprising since CENTURY models decay of SOM as a function of silt and clay content and more C stabilization into passive soil pools as a function of clay content (Metherell et al. 1993). The high-density honeysuckle plot with substantial soil disturbance during interstate construction (HD-SS) sequestered almost as much soil C as the low-density honeysuckle plot with undisturbed soils (LD-U; Fig. 17.4). The soil texture, greater small soil particles (i.e., silt + clay content), explains why the HD-SS plot sequestered high soil C relative to the other two plots (Table 17.2). Despite the low initial soil C in the sub-soils, soil C sequestration was similar to the LD-U plot, even when including the reduction in aboveground inputs to the soil caused by honeysuckle presence. This demonstrates the importance of soil texture in CENTURY estimating soil C stabilization.

While soil texture is an extremely important soil property that controls soil C stabilization in CENTURY, other soil properties, such as bulk density (g soil cm^{-3} soil), also contribute to soil C sequestration. There is a negative linear relationship between bulk density and soil C storage (unpublished data; $y = -233x + 5{,}490$, $r^2 = 0.96$, $p < 0.001$) in CENTURY model simulations. However, this relationship does not have a strong effect on C sequestration. For each 0.3 $g\,cm^{-3}$ increase in bulk density, C storage decreases by almost 70 $g\,m^{-2}$. In contrast, the strength of the silt and clay content on soil C sequestration is greater in CENTURY model simulations. For example, as the fraction of the silt and clay content increases (each 3% increase in silt + clay content) soil C sequestration increases by approximately 90 $g\,m^{-2}$ (unpublished data; $y = -2{,}995x + 3{,}517$, $r^2 = 0.98$, $p < 0.001$). Therefore, more small soil particles (silt + clay fractions) had a larger effect on C sequestration than the higher bulk density in the high-density honeysuckle plot with sub-soil disturbance (HD-SS; Table 17.2). However, bulk density has been shown to have an effect on root growth once bulk density values reach greater than 1.6 $g\,m^{-3}$ (Jim 1998). The addition of this bulk density effect on root production at the 1.6 $g\,m^{-3}$ threshold may need to be incorporated into CENTURY parameterization in order to accurately model urban soil conditions, especially in highly modified soils with intense compaction, like the sub-soil forest plots adjacent to urban interstates. Further CENTURY model modifications may be necessary to correctly simulate the effects of soil disturbances in urban environments.

17.4.3 CENTURY Simulations (2006–2500)

To further investigate C dynamics in the forests adjacent to urban interstates CENTURY simulations were performed until the year 2500 with all natural conditions remaining the same (i.e., no additional disturbances). These long-term CENTURY model simulations were conducted with all the parameters remaining the same as the short-term model runs to determine whether the high-density honeysuckle plot with sub-soils (HD-SS) could store and sequester equivalent tree and soil C to the high-density honeysuckle plot with undisturbed soils (HD-U). The HD-SS plot did surpass the soil C storage in the HD-U plot around year 2177 (after 171 years; Fig. 17.6). As explained in the previous section, this is a result of the CENTURY model's greater importance on soil texture over bulk density in estimating soil C accumulation in forest soils (see Sect. 17.4.2). The soils in the HD-SS plot contained greater silt plus clay content than the HD-U plot, increasing the soil C stabilization in HD-SS as predicted by the CENTURY model. The HD-SS plot and the HD-U plot exhibited similar tree C storage and sequestration. This suggests that C inputs to the soil were similar between the HD-SS and HD-U plots, and that the tree C dynamics were not restrained by the sub-soil disturbance according to the results modeled by CENTURY.

The long-term CENTURY simulations with all model parameters remaining the same as the short-term model runs (e.g., MPR adjustments in place) also predicted whether the high-density honeysuckle plots could store and sequester as much tree and soil C as the low-density honeysuckle plot. Over the 494-year simulation period, the high-density honeysuckle plot (HD-U) stored 23% less tree carbon and 21% less soil C as the low-density honeysuckle plot (LD-U). The HD-U plot did sequester more tree C than the LD-U plot, which is a result of the CENTURY model structure (see Sect. 17.4.2 for explanation). In contrast, the HD-U plot sequestered 58% less soil C as the LD-U plot (Fig. 17.6). These results suggest that the invasive species impact on the tree and soil C dynamics is not alleviated over long time spans as modeled by CENTURY, suggesting the full potential of C storage in these forests will not be attained without the removal of honeysuckle.

The CENTURY simulations with the negative impact of honeysuckle on aboveground production removed from the model runs (adjusted MPR parameter removed) tested whether *honeysuckle removal* from the forest plots would result in similar tree and soil C storage and sequestration to the low-density honeysuckle plot. Once this negative effect of honeysuckle on production was removed from the simulation, both high-density honeysuckle plots (HD-SS and HD-U) sequestered more total soil and tree C than the previous simulations (Figs. 17.5 and 17.6). In fact, the HD-SS plot sequestered more tree and soil C than the low-density honeysuckle plot (LD-U) resulting in equivalent tree C storage and surpassing the soil carbon storage after 264 years (Figs. 17.5 and 17.6). This suggests that high-density honeysuckle plots have the potential to recover C stores similar to forests with low honeysuckle density over long time spans, as long as soil conditions are permissible to C sequestration. However, the high-density honeysuckle plot with undisturbed soils (HD-U) did not attain similar tree and soil C storage to the low-density honeysuckle plot (LD-U) after simulated honeysuckle removal. This suggests honeysuckle presence can have lasting effects on C accumulation rates even after removal from the forest.

These model simulations indicate the potential for adapting CENTURY to urban forests by modeling two aspect of the urban environment that impact forests adjacent to urban interstates, soil disturbance and exotic species. Future work on modeling urban systems using CENURY should focus on simulating other anthropogenic factors such as increased temperatures and pollutant deposition.

17.5 Conclusion

While urban forests alongside interstates are strategically located to provide ecosystem services (i.e., C storage and sequestration), the presence of an exotic invasive shrub reduces the ability of these forests to provide maximal benefits to society. CENTURY predicted that high densities of honeysuckle reduced tree and soil C sequestration, suggesting the future ability of these forests to act as a C sink is threatened by honeysuckle presence. Removal of honeysuckle may improve the soil C storage and sequestration in these forests. However, future stressors and/or disturbances will change the trajectory of carbon storage and sequestration.

The CENTURY model structure was originally developed to simulate C accumulation for natural ecosystems (i.e., grasslands, forests, savanna). Thus, the carbon sequestration predictions for forests adjacent to urban interstates are estimated using natural soil forming processes when human disturbances and stresses are unknown. Understanding site history for ecosystems in urban environments is extremely important when modeling in CENTURY, which may still estimate C stabilization with more natural conditions than are occurring in urban environments even with inclusion of human disturbances in model simulations.

Future work on using CENTURY to model C dynamics in urban environments and specifically forests adjacent to urban interstates should include model runs that simulate varying scenarios of human and natural disturbance, which will decrease the importance of natural controls estimating C dynamics in CENTURY model runs. The next phase of CENTURY modeling work on urban forest interstates will include simulating additional threats caused by human interactions with these forests (e.g., increased temperature). This type of predictive modeling can provide information to land managers as to the best possible practices for forest carbon storage and sequestration. A relevant suggestion to those managing forests adjacent to urban interstates in Louisville, KY would be to consider the removal of Amur honeysuckle. Forests alongside urban interstates are extremely important in providing ecosystem services (i.e., C storage and sequestration) in the urban landscape and should be managed to maintain these benefits to society.

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Part V
Sustainable Management of Urban Ecosystems

Chapter 18
Global Urbanization and Demand for Natural Resources

Chia-Tsung Yeh and Shu-Li Huang

Abstract The global urban population is now increasing at an unprecedented rate, creating tremendous stress on local, regional, and global environments and natural resources. Over half of humanity currently lives in cities and this proportion is expected to increase to nearly 60% over the next 40 years. Urbanization is changing the way humans consume natural sources and is transforming land use. This chapter investigates global urbanization and increased urban demand for natural resources. Land use changes introduced by global urbanization and their impacts on environments and natural resources are explored. Concepts and methodologies for framing global urbanization and demand for natural resources, such as urban metabolism, ecological footprint, lifecycle analysis, and their recent applications are reviewed. The importance of innovative planning and management of urban development to minimize the resource demands of cities by reducing energy and material inflows and closing the urban metabolism loop is discussed.

Keywords Urbanization • Natural resources • Developing countries

List of Abbreviations

GHG	Greenhouse gas emissions
IEA	International Energy Agency
LCA	Life cycle assessment
NPS	Non-point source

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OECD Organization for Economic and Co-operation and Development
UHIs Urban heat islands
UN United Nations

18.1 Introduction

Today we face the great challenge of sustainability on a planetary scale, brought about by the confluence of two global trends: transition toward urbanization and global environmental change (Seto and Satterthwaite 2010). With urban areas accounting for more than 70% of total energy consumption] and a corresponding proportion of the world's carbon dioxide emissions, when aggregated globally, local processes in urban areas have the potential to adversely affect the Earth's systems (Grimm et al. 2008).

Since 1950, the proportion of the world's population living in urban areas has increased from 13% to more than 50% (UNPD 2011). Humanity is now entering a new world, a world of the city, by the city, and for the city. Urbanization will be the main obstacle to, and opportunity for, sustainable development (McDonald 2008). Although the total urban area worldwide remains a relatively small fraction of the Earth's terrestrial surface, urbanized areas account for roughly 75% of the global consumption of resources (Angel et al. 2005; Pacione 2009).

The concentration of people in cities inevitably increases the concentration of pollution, often exceeding the capacities of natural ecosystems to absorb and assimilate pollutants. Their impact has also been global in terms of greenhouse gas emissions (GHG) and resources consumed (Grimm et al. 2008). Urbanization will also adversely affect ecosystem services and biodiversity. For example, the spatial concentration of humanity in cities results in partial or complete appropriation of freshwater for urban use (McGranahan et al. 2006; McDonald 2008). Urban expansion and the demand for resources and energy are transforming the Earth's terrestrial ecosystems. Urbanization and urban development is driving habitat degradation and species extinction, changes in biogeochemistry, and modifying hydrological systems (Seto and Satterthwaite 2010).

Not only is the world's urban population increasing, but the area extent of individual cities is dramatically growing in size (Grimm et al. 2008). Cities have significant influences on natural ecosystems, and these impacts have become increasingly apparent as the size and number of cities continues to grow. With these urban changes the intensity and extent of urban impacts on ecosystems continues to increase (Hu et al. 2009). The clearing of land for cities and roads and the demand for goods and services by urban residents are the major drivers of regional land use and land cover change (Batisani and Yarnal 2009).

Although urbanization is typically viewed as a demographic and economic phenomenon, it is also a process of ecological transformation, significantly influencing how local and global ecosystems function. Consequently, urbanization is a key driver of global environmental change. Moreover, urban residents generally depend on ecosystem services provided by areas extending well beyond their city's boundaries and are often ten to hundreds of times the city. Expanding cities tend to appropriate a disproportionate share of ecosystems when measured in resource inputs and waste sinks (Güneralp and Seto 2008; Santoro et al. 2009; Hubacek et al. 2009; Suthar et al. 2009).

Studies of human and natural systems that define the current understanding of resiliency of urban areas suggest that cities are the major sources of global environmental problems as well as being hot spots for solutions (Cousins et al. 2007; Liu et al. 2007; Grimm et al. 2008). Issues and challenges to global environmental sustainability by urbanization have been extensively discussed in literature, including specific challenges posed by Asian urbanization (Ooi 2009), urban land use trends, climate impacts (Seto and Shepherd 2009; Grimmond 2007), and challenges associated with clean air (Parrish and Zhu 2009). This chapter focuses on the development of global urbanization and the demand for natural resources associated with increased urbanization. To respond to the increasing urbanization in developing countries and critical issues in the twenty-first Century, most cases examined in this chapter are from developing countries.

18.2 Global Urbanization

Cities offer significant opportunities for economic and social development. In fact, cities have been focal point for economic growth, innovation, and employment throughout history. With rapid economic development and population growth, urbanization is taking place at an unprecedented pace worldwide. In 2007, humanity crossed a momentous milestone: for the first time in human history, more than half of the world's population was living in cities. The global urban population was 3.4 billion in 2009 and is predicted to increase to 6.3 billion by 2050 (UNPD 2011).

The best source of quantitative global data for urbanization is the United Nations (UN) Population Division. Since 1988, the Population Division of the UN Department of Economic and Social Affairs has issued global population data every 2 years and revised and updated estimates and projections of urban and rural populations and their major urban agglomerations in all countries. Table 18.1 shows the urbanization development and prospects worldwide and major regions. These data indicate that high urbanization rates prevail in North America (82.1%), Europe (72.8%), and Latin America and the Caribbean (79.6%). However, Africa (40%) and Asia (42.2%), particularly countries in Southeast Asia such as China (46.1%) and India (29.7%), are just beginning their shift toward urbanization and will likely undergo rapid urbanization during the next several decades. It also shows that population growth and urban population growth in particular are predicted to be extremely rapid in the developing world, averaging 2.3% annually during 2010–2050.

Urbanization characterized economic development in Europe, North America, and Japan during the nineteenth and early twentieth centuries, and led to a continuous

Table 18.1 Urbanization development and prospects, worldwide and in major regions, 1950–2050 (UNPD 2011)

	Population in urban area (millions)			Urban share (%)
	1950	2000	2050	2010
World	729	2,837	6,286	50.5
More developed regions	427	869	1,100	75.2
Less developed regions	302	1,968	5,186	45.1
Least developed countries	15	167	914	29.2
Africa	33	295	1,231	40
Asia	229	1,361	3,382	42.2
Europe	281	514	582	72.8
Latin America and the Caribbean	69	393	648	79.6
Northern America	110	252	404	82.1
Oceania	8	22	38	70.2

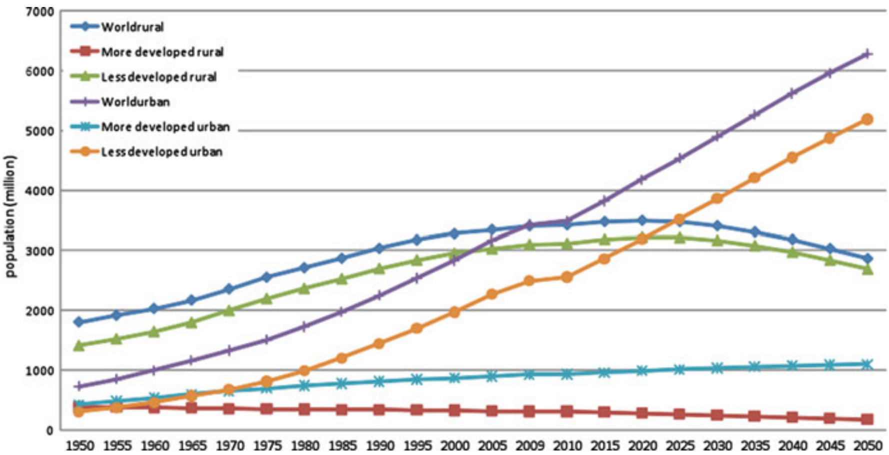

Fig. 18.1 Global urbanization development and prospects (UNPD 2011)

economic, demographic, and functional growth. Increasingly urbanization in developing countries became an important issue in the second half of the twentieth Century, when most countries in Asia and Africa regained independence.

This study adopts the UN definitions of more and less developed countries and least developed countries to investigate the evolution of populations in urban areas during 1950–2050 (UNPD 2011). Clearly, future urban growth will mainly occur in less developed countries. According to UN projections, more than 80% of the world's urban population in 2050 will live in less developed countries. Within this group of less developed countries, African and Asian cities will experience the higher urban growth than before over the next 40 years. Figure 18.1 shows the future role of less developed regions with regard to urban population. This figure provides some evidence that the worldwide urban growth is mainly driven by the less developed countries.

Table 18.2 Population and the world's megacities, 2010–2025 (UNPD 2011)

	2010		2025	
Rank order	Urban agglomeration	Population (millions)	Urban agglomeration	Population (millions)
1	Tokyo	36.67	Tokyo	37.09
2	Delhi	22.16	Delhi	28.57
3	São Paulo	20.26	Mumbai (Bombay)	25.81
4	Mumbai (Bombay)	20.04	São Paulo	21.65
5	Ciudad de México (Mexico City)	19.46	Dhaka	20.94
6	New York-Newark	19.43	Ciudad de México (Mexico City)	20.71
7	Shanghai	16.58	New York-Newark	20.64
8	Kolkata (Calcutta)	15.55	Kolkata (Calcutta)	20.11
9	Dhaka	14.65	Shanghai	20.02
10	Karachi	13.12	Karachi	18.73
11	Buenos Aires	13.07	Lagos	15.81
12	Los Angeles	12.76	Kinshasa	15.04
13	Beijing	12.39	Beijing	15.02
14	Rio de Janeiro	11.95	Manila	14.92
15	Manila	11.63	Buenos Aires	13.71
16	Osaka-Kobe	11.34	Los Angeles	13.68
17	Al-Qahirah (Cairo)	11.00	Al-Qahirah (Cairo)	13.53
18	Lagos	10.58	Rio de Janeiro	12.65
19	Moskva (Moscow)	10.55	Istanbul	12.11
20	Istanbul	10.52	Osaka-Kobe	11.37
21	Paris	10.49	Shenzhen	11.15
22	Seoul	9.77	Chongqing	11.07
23	Chongqing	9.40	Guangzhou, Guangdong	10.96
24	Jakarta	9.21	Paris	10.88
25	Chicago	9.20	Jakarta	10.85
26	Shenzhen	9.01	Moskva (Moscow)	10.66
27	Lima	8.94	Bogotá	10.54
28	Guangzhou, Guangdong	8.88	Lima	10.53
29	Kinshasa	8.75	Lahore	10.31
30	London	8.63	Chicago	9.94

Table 18.2 identifies worldwide emerging megacities will have over ten million inhabitants by 2050. In 2010, 21 cities had over ten million inhabitants, 16 of which were in less developed countries. By 2025, the number of megacities is expected to increase to 29, 24 of which will be in less developed countries. Many cities in Pacific Asia, for instance, have experienced dramatic economic growth, and this region is now integrated into the global economy. Numerous reports suggest that these increases in urban populations generate serious environmental and social problems and accelerate global environmental change (Bengtsson et al. 2006; Grimmond 2007; Parnell et al. 2007).

18.3 Urbanization and Natural Resources Consumption

Urbanization is typically characterized by a large human population combined with increased per capita energy and resources consumption as well as extensive landscape modifications (Luck and Wu 2002; Liu 2009). Urbanization is dependent on a steady supply of natural resources, including freshwater, fuel, land, food, and raw materials (Hardoy et al. 2001; Chen 2007). Humanity's demand for natural resources far exceeds what the planet can provide sustainably, and this over-consumption is set to increase significantly due to rapid economic expansion and urbanization (Hoekstra and Chapagain 2007). Dramatic increases in consumption of natural resources generally accompany rapid urbanization.

18.3.1 Water

Of the Earth's 1,386 million cubic kilometers of water, only 2.5% of that quantity is fresh water and nearly one-third of this smaller amount is available for human use (Postel et al. 1996). Over half of available freshwater supplies are already used for human activities and this proportion is increasing in response to growing agricultural, industrial, and residential demands (Postel et al. 1996; Vorosmarty and Sahagian 2000). Freshwater availability varies with precipitation and evaporation/transpiration rates, and these rates are affected by climate, topography, vegetation cover, and soil characteristics (Jenerette et al. 2006).

Accompanying rapid increases in global urbanization is growth in urban water use by both industrial and domestic sectors (Gleick 2003; Jury and Vaux 2005). The growing demand for water poses significant challenges for water resource management in both developed and developing nations (Jenerette and Larson 2006; Biwas 2006). Water management is a critical issue in megacities (Varis et al. 2006) as water is intricately linked to not only domestic and industrial use but also adequate sanitation, and protection from natural disasters. However, over one billion people worldwide lack access to clean and safe water and 2.4 billion people live without adequate sanitation (Cain and Gleick 2005; Grimm et al. 2008).

The scarcity and sustainability of renewable water resources is affected by four comprehensive factors: (1) population, (2) per-capita water use, (3) climatic change, and (4) allocations for water conservation. These factors affect urban water use both individually and interactively (Jenerette and Larson 2006). These factors affect urban water use both individually and interactively. With increasing populations, cities are increasing water requirements for concentrated areas. Moreover, changes to global and regional climates are predicted to affect the total amount of water available and variations in water resources worldwide (Tudhope et al. 2001).

The impact of rapid urbanization on water demand is sizable. Urbanization increases a nation's water demands much faster than they would merely as a function of population growth. Global urban water use increased by a factor of

20 during 1900–2000 as the world's population only increased by a factor of 3 (Bao and Fang 2007). Continuing this trajectory would leave 55% of the world's population in water crisis by 2050 (Song et al. 2004). For instance, domestic water demand due to urban growth and living standards has led to water shortages in China. Water crises existed in over 400 Chinese cities in 2000 (Zhang et al. 1992; Zhu et al. 2001). Further, rapid urbanization usually results in rapid degradation of water quality.

The freshwater supply is a particularly important ecosystem service that is intensively managed in almost all populated regions worldwide (Vorosmarty et al. 2003; Meybeck 2004). In many regions, freshwater supplies are threatened by local consumption patterns of both agriculture and urban infrastructure, industry, and residents (Jenerette et al. 2006). Urban demand for freshwater is also spatially heterogeneous due to the locations of cities, variations in per capita usage, and variations in urban populations. The mismatch between a location and its water supply is the primary cause of urban water shortages. Consequently, effective water management in megacities is essential to megacity survival. Tortajada (2008) analyzed the delicate relationship between economic development and urban natural resource management. His analysis highlighted the diminishing freshwater supply for Mexico City and the associated sanitation problems that are now faced by many rapidly developing cities.

Competition for water, frequently associated with competition for access to land, is typically exacerbated in peri-urban areas (Ducrot et al. 2004). Moreover, agriculture and urban industrial sectors often compete for water (van Beek et al. 2010). For instance, increased competitions for water under drought conditions and rapid population growth in Southern California have resulted in intense competition for water among different sectors. Moreover, transferring water from agricultural to other uses has generated political and economic conflict among different stakeholders (Knapp et al. 2003). In developing countries, urban water management is a much more complex and interdisciplinary issue than in developed countries. A new water management paradigm as a future strategy and policy for water resources management should integrate institutional, social, economic, and environmental perspectives. (Vo 2007).

Moreover, a rapid concentration of people in numerous urban coastal cities has resulted in excessive groundwater withdrawal, causing large-scale problems such as land subsidence and salinization. For instance, on the Kanto plain, the largest depositional plain in Japan and home to the Tokyo Metropolitan Area, excessive groundwater withdrawal has resulted in human-induced disasters such as land subsidence (Hayashi et al. 2009). Nowadays, other large coastal cities in Asia, such as Shanghai, Bangkok, Manila, and Jakarta, have also experienced similar problems caused by excessive groundwater withdrawal (Hayashi et al. 2009).

Notably, many desert cities rely on infrastructure that consumes considerable amounts of energy to supply water from large hydraulic hinterlands. These hinterlands often compete for scarce water with other cities, the environment, and agriculture (Ezcurra 2006; Gober 2010).

18.3.2 Agricultural Products

The global demand for agricultural products is increasing rapidly due to population growth and shifting consumption patterns (van Beek et al. 2010). Food production will need to increase dramatically to meet the projected worldwide demand. The task of increasing food production will be more complex due to the interactive effects of changes in climate, atmospheric composition, land use, and other drivers of global change (Gregory and Ingram 2000). Unfortunately, today's rapidly urbanizing world is facing adverse environmental impacts and the conversion of agricultural land into developed uses, which are the major problems resulting from rapid urbanization in most urbanized countries and regions (OECD 2001). Accelerating urbanization also degrades agricultural landscapes, which can adversely affect various ecological processes, finally threatening regional sustainability (Fu et al. 2006).

Urbanization frequently leads to the conversion natural landscapes into urban landscapes and intensifies competitions between different land-use practices in space and time, thus directly and indirectly affecting soil resources and food security. Urbanization is now considered the primary threat to future agricultural production due to the increasing risk of soil pollution through urban waste disposal and acid deposition derived from urban air pollution. For example, processes of rapid urbanization, industrialization, and the accompanied agricultural restructuring in China since the 1990s have resulted in the loss of massive amounts of farmland to the benefit of market farming and non-agricultural development (Ren et al. 2003). Although China feeds 22% of the global population, it has less than 9% of the world's cultivated land. Increasing concern over land scarcity in China is expressed in terms of soil availability for agricultural production, which has decreased due to rapid population growth and accelerated urbanization and industrialization over the last two decades (Yang and Li 2000; Lin and Ho 2003; Chen 2007). To feed its 1.3 billion people with per capita cultivated land far below the world average, China is already facing the great challenge of land scarcity.

18.3.3 Energy

Urbanization has significant implications for urban energy demand by affecting the urban infrastructures. Concentrated economic activity results in land scarcity and the need to erect multi-level buildings (Parikh and Shukla 1995). Such development directly and indirectly impacts energy consumption (Madlener and Sunak 2011). Growing cities typically have great demand for energy-intensive products. Generally, construction of urban infrastructure, including roads, bridges, office buildings, sewage lines, and power plants, is associated with a high energy input (Jones 2004). Moreover, usage and maintenance of urban infrastructure, such as sewage networks, lighting, and water and waste treatment facilities, require additional energy.

Table 18.3 World energy demand in cities by fuel in the reference scenario (IEA 2008)

	2006		2015		2030		
	Mtoe	Cities as a % of world	Mtoe	Cities as a % of world	Mtoe	Cities as a % of world	Average annual growth rate 2006–2030
Coal	2,330	76%	3,145	78%	3,964	81%	2.2%
Oil	2,519	63%	2,873	63%	3,394	66%	1.2%
Gas	1,984	82%	2,418	83%	3,176	87%	2.0%
Nuclear	551	76%	630	77%	726	81%	1.2%
Hydro	195	75%	245	76%	330	79%	2.2%
Biomass and waste	280	24%	358	26%	520	31%	2.6%
Other renewables	48	72%	115	73%	264	75%	7.4%
Total	7,908	67%	9,785	69%	12,374	73%	1.9%
Electricity	1,019	76%	1,367	77%	1,912	79%	2.7%

According to World Energy Outlook 2008, roughly two-thirds of the world's energy (an estimated 7,900 Mtoe in 2006) is consumed by cities, even though only roughly half of the world's population lives in cities. Urban residents consume more coal, gas and electricity but less oil than the global average (International Energy Agency (IEA) 2008). Increased urbanization through to 2030 is projected to drive up urban energy use to almost 12,400 Mtoe (Table 18.3). By 2030, cities will account for 73% of worldwide energy usage. Some 81% of this projected increase in energy usage by cities during 2006–2030 will be from countries that are not members of the Organization for Economic and Co-operation and Development (OECD). The scale and patterns of urban energy use have significant implications both energy security and global GHG emissions. Alert to climate change, some cities are actively seeking to reduce energy usage and CO_2 emissions.

18.3.4 Timber and Forestry

Urban areas also have considerable demand for timber for heating, settlements, recreation, tourism, all of which exert pressures on forests. Loss of the world's tropical forests is positively correlated with urban population growth and exports of agricultural products (DeFries et al. 2010). For instance, approximately one-third of the world's mangrove forests no longer exist. Destruction of mangrove forests is typically positively correlated with population density. Moreover, a major reason for mangrove forest destruction is urban development (Alongi 2002).

Even though forests in metropolitan areas are increasingly acknowledged for their role in improving the quality of the urban environment, rapid urbanization is impacting their existence, structure and health. Recent research assessed how the rapid urbanization process in Turkey impacts forests (Atmis et al. 2007).

18.4 Urban Land Use Change and Its Impacts on Natural Resources

Urbanization and land consumption have become contentious issues in public debate and academia (Antrop and Van Eetvelde 2000). Although changes in land use have allowed humans to appropriate an increasing share of resources, they often undermine the capacity of ecosystems to sustain food production, provide freshwater and regulate climate and air quality (Langpap and Wu 2008; Foley et al. 2005). Material demands of economic production and human consumption have resulted in dramatic changes in land use and cover, biodiversity, and hydrologic systems both locally and regionally.

Urbanization has resulted in irreversible effects on the structure, function, and dynamics of ecological systems across a wide range of spatial scales due to loss of arable land, habitat destruction, and decline in natural vegetation cover (Luck and Wu 2002). Many studies have examined land use changes due to rapid urbanization in countries such as in China and India (Wu et al. 2006; Quan et al. 2007; Dewan and Yamaguchi 2008; Liu et al. 2010). Schneider and Woodcock (2008) compared the urban form and growth of 25 mid-sized cities with different geographical settings and levels of economic development. Urban sprawl is considered a major force driving land use and land cover change. Additionally, changes in urban landscape patterns adversely affect ecological integrity; and these processes are tightly intertwined with the mosaic of landscape elements resulting from urbanization (Pino et al. 2000; Alberti et al. 2003). A recent study of the Phoenix Metropolitan Region shows that urbanization-induced changes in atmospheric CO_2, N deposition, and air temperature changes significantly impact urban ecosystem functioning (Shen et al. 2008).

Further, the conversion of a natural landscape into an urban landscape directly and indirectly influences soil resources, water resources, and ultimately food security (Alphan 2003; Chen 2007). In developing countries, rapid urbanization and industrialization have diverted large amounts of arable land for industrial production, housing, and infrastructure (Liu et al. 2010). This consumption of arable land undermines ecological sustainability directly through land-use conversion (Liu et al. 2008). Morello et al. (2000) examined conflicts between agriculture and urban development in Pampa Ondulada, the ecological region in which the city of Buenos Aires is located, which is one of the world's richest and most productive agricultural areas. It describes the ecological changes brought by urban growth in peri-urban and rural areas and includes an analysis of the changes and the effect on ecological services.

Haase (2003) examined the per capita consumption of land resource and impacts associated with urban sprawl. More recently, He analyzed the impact of urban land use change and land surfacing on the long-term urban water balance in Leipzig, Germany (Haase 2009). His long-term observations of urban sprawl showed that the key issue is the cumulative impact of land use change and surface sealing, rather than short-term consequences that are likely to impair the urban water balance. Moreover, urban expansion is also a major driving force altering local and regional hydrology and increasing non-point source (NPS) pollution. Tang et al. (2005)

examined land use changes in the Muskegon, US, and assessed the impacts of land use changes on long-term runoff and NPS pollution using an environmental impact analysis model. Moreover, their research indicates that the watershed would likely be subjected to impacts from urbanization on runoff and some types of NPS pollution.

Land consumption for urbanization frequently drives other environmental changes (Grimm et al. 2008). For instance, high-density development can create urban heat islands (UHIs), increasing urban temperatures and likely increasing water consumption and energy use. One mitigation strategy touted by many urban planners is to plant trees and irrigated vegetation to prevent daytime heat storage and facilitate cooling, even though this strategy requires water resources already under pressure (Gober et al. 2010). Gober et al. (2010) investigated the tradeoff between water use and nighttime cooling in Phoenix, USA. They found that increasing irrigated landscaping lowered nighttime temperatures caused by UHIs.

From the landscape ecology perspective, urban land use often has countless effects on landscape ecology (Grimm et al. 2008). Urbanization can fragment natural habitats; simplifies species composition; disrupt hydrological systems; significantly alter energy and material flows; and change nutrient cycling in ecosystems (Grimm et al. 2000; Alberti et al. 2003).

Since peri-urban areas have unique ecological characteristics, conversion of these areas generally influences ecosystem functioning and services. Notably, land cover and land use changes in peri-urban areas are often not considered important by rural and urban administrators (Douglas 2006). Simon (2008) analyzed current thinking about the environment-development opposition in peri-urban fringe areas and underscored concerns about the real-world limitations of traditional concepts of a simple rural-urban dichotomy. The loss of ecosystem services resulting from peri-urbanization has increased the demand for natural resources from areas far from these urbanized areas. For instance, while investigating urbanization in the Mexico City, Aguilar (2008) determined that peri-urbanization has resulted in marked environmental damage, and identified a lack of effective planning and urban governance for sustainable development in the city's preservation zone.

18.5 Conceptual and Methodological Development

Research leading to the development of concepts and methodologies that assess and natural resource use by cities are important for documenting the environmental burden of urbanization. Concern over the impacts of urban life in the global context has given rise to a focus on the Ecological footprint of cities. Rees (1992) introduces the concept of ecological footprint as a measure of the area of biologically productive land and sea required to produce the renewable resources with respect to how population consumes and assimilates the waste it generates. The management of a city's resource metabolism, including natural capital supporting these flows, is becoming a central concern for cities moving toward sustainability (Wackernagel and Rees 1996).

Through trade and natural flows of ecological goods and services, urban areas appropriate the carrying capacity of distant ecosystems in terms of resource inputs and waste sinks (Alberti 2005). Restated, urban dwellers depend on the productive and assimilative capacities of ecosystems well beyond their city boundaries to produce flows of energy, material goods, and nonmaterial services. By inverting the standard carrying capacity ratio, ecological footprint determines the total amount of land required to provide resources and energy for a given population with a specified living standard. This ecological footprint has been applied extensively to assist cities and regions in managing their ecological assets, and supporting their sustainability efforts (Wackernagel et al. 2006).

Alternative formulations for footprint accounting have been developed to examine urban dependency on ecosystems (Luck et al. 2001; McDonald and Patterson 2004). Using the ecological footprint methodology, Jenerette et al. (2006) examine production and consumption patterns of freshwater for selected cities in China and the United States. Furthermore, the authors address the questions concerning the relationship between social and ecological systems and identify the constraints to the sustainable withdrawal of water resource.

Urban metabolism was first suggested by Wolman (1965) as the materials, energy, and food supplies brought into cities, transformed within the cities, and the products and wastes then sent out from the cities. This concept underscores the fact that a metabolic cycle is not complete until waste from daily consumption has been removed and disposed of adequately with minimum nuisance and hazard to life. Extending from the concept of circular metabolism, Huang et al. (1998) developed a framework of urban ecological economic system and selected 80 indicators to measure the urban sustainability of Taipei, Taiwan. The metabolism of cities has been studied extensively worldwide. Kennedy et al. (2007) reviewed eight urban metabolism studies conducted since 1965 and identified metabolic processes that threaten the sustainability of cities. The major environmental problems and associated social costs of an urban ecosystem are related to rapid increases in the resource inputs needed to satisfy urban consumption and for the disposal of construction waste, both of which are nuisances to urban dwellers.

Life cycle assessment (LCA) has also matured as an analytical method to the point where it has been used to assess the current metabolism of cities. Mora (2007) explored the relationship between life cycle and sustainability of engineering works and the impact of urban growth and urban infrastructure on the environment via the consumption of raw materials and energy. Analyses of urban metabolism have largely quantified material inflows and outflows. Transformation of materials into economic assets that sustain an urban metabolism and maximize their usefulness for human societies must be also be analyzed. However, measuring material flows does not allow for comparisons of the qualitative usefulness of different materials to socioeconomic systems (Huang and Hsu 2003; Huang et al. 2006). Aggregating material flows according to their mass neglects the relative contribution associated with values of materials with different qualitative contents. Huang and Chen (2009) applied Odum's energy concept to integrate energy and material flows in a study of the socioeconomic metabolism of Taipei and its surrounding areas. Urban sprawl in Taipei was also considered in studying its relationships with the changing socioeconomic metabolism of the region.

The idea of the Ecocity was first proposed by Register (2002) and focused on minimizing the required inputs of energy and resources and outputs of waste. Many examples of eco-town and city initiatives now exist. One response to reduce the impact of increasing urbanization on the loss of ecosystem services is to minimize the spatial extent of built-up lands by promoting compact urban forms.

18.6 Conclusions

Although cities are typically the locus of economic growth for most nations, they cause sustainability problems for an increasingly urbanized world. Despites positive causal relationships between urbanization and the demand for natural resources, there are also ways to alleviate the increasing burden of urbanization on natural resources systems. Many urban planners are aware of the burden on natural resources from urbanization and stress the importance of reducing this demand as a local response to global environmental change. If well managed, cities with a high population density may minimize the effect of man on natural ecosystems.

Many innovative ideas regarding the way a city could be planned or developed to minimize natural resource demand have been proposed. Considerable theoretical debate and discussion in various disciplines have focused on the practical challenges of the compact city paradigm (Jenks et al. 2000; Couch and Karecha 2006). Measures and indices have been developed to measure compactness and sprawl to generate empirical evidence to advance this debate (Burton 2002; Frenkel and Maya 2008). In addition to spatial strategies, some urban areas have used economic tools to curtail resource use.

Urban development faces a number of obstacles and constraints when implementing these innovative concepts and models, especially in developing countries. Although there is an urgency to assist cities in their efforts to incorporate new concepts and ideas of sustainable urban development into practice, it is still worth stressing that there is a vital need to reduce demands for natural resources. The continued development and application of methodologies for studying the intervention of urbanization on natural resources remains essential.

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Chapter 19
Towards Greening of Urban Landscape

Rattan Lal

Abstract Rapid urbanization since 1950, both in developed and developing countries, has led to a large-scale conversion of prime agricultural/farm lands to build up areas under concrete and asphalt. The associated disturbance of the ecosystem structure and functions leads to disruption in cycling of elements (C, N, P), perturbation of the hydrologic cycle and the energy budget, and depletion of the ecosystem C pool. The global rate of urbanization is about 2 Mha/year. Land area under urban centers is estimated at about 24 Mha (2.6% of the total land area) in the U.S.A. Urban centers consume a large proportion of total energy, transport fuel, food, water and other consumable, and are thus principal sources of gaseous emissions. Nonetheless, the green areas within the urban centers (i.e., home lawns, turfs, urban forests, and urban agriculture) have the technical potential to off-set some of the gaseous emissions. This strategy is to enhance the net ecosystem C budget of the green area by reducing the hidden C costs of inputs, enhance above and below ground biomass production, and restore abandoned lands and buildings for agricultural production. Sustaining agronomic production implies improving soil structure and tilth, alleviating soil compaction, creating favorable water and nutrient regimes, and increasing soil organic C pool and especially its depth distribution in the sub-soil.

Keywords Urban encroachment • Urban ecosystems • C sequestration • Climate change • Green roofs

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List of Abbreviations

C	carbon
GCC	global carbon cycle
GHGs	greenhouse gases
HCC	hidden carbon costs
INM	integrated nutrient management
IPM	integrated pest management
NPP	net primary productivity
SOC	soil organic matter
UA	urban agriculture

19.1 Introduction

The rapid urbanization started at around 1950. The urban population increased in all regions, but especially in the developing countries (Table 19.1). Increase in urban population (millions) between 1950 and 2007 was from 224 to 967 in Africa, 1,411 to 4,030 for Asia, and 168 to 572 for Latin America and the Caribbean. By 2050, the population will increase (millions) to 2,000 in Africa, 5,266 in Asia, and 769 in Latin America, when the urban population as percent of total population will be 62 for Africa, 66 for Asia, and 89 for Latin America (Table 19.1). The rate of urbanization between 1950 and 2007, and between 2007 and 2050 has been in the order of Africa>Asia>Europe>Latin America (Table 19.2). Although, rapid rate of urbanization started in more developed countries (i.e., USA, Europe), the present rate of urbanization is greater in developing than developed countries (Table 19.2). Among the developing regions, Latin America and the Caribbean has the high level of urbanization (U.N. 2008). The debate over the impact of urbanization on land use and ecosystem functions is especially important to Latin America and the Caribbean. This region has the highest rates of urbanization growth in the world (Cohen 2004; Rodriquez and Bonilla 2007).

There are numerous consequences of rapid urbanization (see Chap. 1). An important among these is change in land use and land cover, including deforestation. In general, most of the urban encroachment happens on prime farmland. Vining et al. (1977) observed that urban encroachment occurs on land that is ideal for agriculture (or prime farm land). Such prime land is finite and limited in supply world wide. Similar observations were made in Chicago, USA, by Greene (1997) who conducted a survey of urban encroachment between 1975 and 1990. He observed that "edge cities" or urban fringe of low-density residential uses are replacing farmland at a rapid rate. Greene reported that most large cities of the Midwest USA reveal a similar pattern of replacing prime farmland on the urban fringe.

Although the total area under urban land is small (i.e., in Australia, Canada, New Zealand and USA) (Table 19.3), its impact is greatly accentuated because of the

Table 19.1 Trends in urban population on continental basis (Recalculated from U.N. 2008)

	Total urban population (10^9)					Urban population as (%) of total				
Region	1950	1975	2007	2025	2050	1950	1975	2007	2025	2050
Africa	0.224	0.416	0.965	1.394	1.998	14.5	25.7	38.7	47.2	61.8
Asia	1.411	2.394	4.030	4.779	5.266	16.8	24.0	40.8	51.1	66.2
Europe	0.548	0.676	0.731	0.715	0.664	51.2	65.7	72.2	76.2	82.8
Latin America and the Caribbean	0.168	0.325	0.572	0.688	0.769	41.4	61.1	78.3	83.5	88.7
North America	0.172	0.243	0.339	0.393	0.445	63.9	73.8	81.3	85.7	90.2
Oceania	0.013	0.021	0.034	0.041	0.049	62.0	71.5	70.5	71.9	76.4
Total	2.536	4.075	6.671	8.010	9.191					

Table 19.2 Rate of urbanization in different regions of the World (Adapted from UN 2008)

Region	Rate of urbanization (%) 1950–1975	1975–2007	2007–2025	2025–2050
Africa	2.28	1.28	1.10	1.08
Asia	1.42	1.66	1.24	1.04
Europe	1.00	0.29	0.30	0.38
Latin America and the Caribbean	1.56	0.78	0.36	0.24
North America	0.58	0.30	0.29	0.20
Oceania	0.57	–0.05	0.11	0.24

Table 19.3 Estimates of urban land area as a percent of total land area for some countries in Europe and North America (Adapted from Demographia 2010)

Country	Urban land area (% of total area)
Australia	0.25
Canada (all)	0.27
Canada (agric. belt)	3.29
France	12.38
Germany	27.53
Great Britain	5.96
Italy	20.09
Japan	14.29
Netherlands	28.28
New Zealand	1.42
Switzerland	17.54
USA	2.63

conversion of prime farmland. Historically, as early as 1980, the impact of urbanization on food price inflation and agricultural shortages have been felt across the world (Plaut 1980), but has exacerbated during the twenty-first century. A drastic spike in world's food prices, due to a wide range of interacting factors including conversion of food staples to biofuels and increase in frequency of extreme events (i.e., drought) relate to climate change, in 2008 increased the number of food-insecure population from 925 million to 1,020 million (FAO 2010). It is also estimated that additional 44 million people world wide have fallen below the poverty line since 2010 due to increase in food prices.

Increase in the demand for natural resources is another inevitable consequence of urbanization in both developed and developing countries. In addition to being a socioeconomic phenomenon, urbanization is also an anthropogenically-driven process of drastic ecologic transformation. Such a transformation leads to increase in consumption of natural resources including land, water, forest, energy and minerals. The demand on natural resources increases with increase in number and size of megacities which disproportionately appropriate the limited resources (Huang et al. 2010).

The objective of this chapter is to identify researchable priorities which minimize conversion of prime farmland to urbanization, sequester C in urban lands, and reduce the net emissions of greenhouse gases (GHGs) from urban ecosystems.

19.2 Land Use Conversion by Urbanization and Impact on Natural Resources

The global urban expansion rate is estimated at 2 million ha (Mha) per year (Holmgren 2006; Table 19.4). Encroachment includes 1.6 Mha on cropland, and 0.2 Mha on each of forest and grasslands (Angel et al. 2005). Land area needed to accommodate and provide infra-structure to one million people is about 40,000 ha. With an annual global population growth of 75 million, land area converted to urban use is ~3 Mha/year. The area needed for urbanization is much smaller in developing than developed countries, thus an estimate of 2 Mha by Holmgrsen is realistic.

The data in Table 19.5 show the temporal changes between 1945 and 2002 in area under urban land use in USA, the U.S. Corn Belt region, and in the state of Ohio. The land area under urban use is 24 Mha in USA, 3.2 Mha in the Corn Belt and 1.0 Mha in Ohio (Table 19.5). With total land area of 916 Mha, that under urban use in the U.S. is about 2.6% (Table 19.3). Temporal change in urban land area in the U.S., indicate a linear increase between 1954 and 1997 (Fig. 19.1).

Such a rapid rate of urbanization has caused irreversible change in structure, functions and dynamics of ecosystems (Luck and Wu 2002; Huang et al. 2010). Such transformations are equally severe in developing countries such as China

Table 19.4 Global land use conversion to urbanization (10^6 ha) in 2000 (Adapted from Angel et al. 2005, Holmgren 2006)

From \ Into	Forest	Woodland/grassland	Agricultural crops	Urban
Forest	**3,790**	3.0	9.8	0.2
Woodland/grassland	1.4	**3,436**	1.0	0.2
Agricultural crops	4.3	2.0	**1,514**	1.6
Urban	NA	NA	NA	**38.0**

Table 19.5 Land in urban areas in the U.S.A. (Adapted from ERS-USDA 2010)

	Area (10^6 ha)		
Year	USA	Corn belt	Ohio
1945	6.08 L[1]	1.07	0.33
1949	7.40 L[1]	1.19	0.36
1959	11.01	1.76	0.61
1969	12.56	2.05	0.68
1978	18.08	2.56	0.82
1982	20.32	2.72	0.87
1992	23.85	3.09	0.98
2002	24.12	3.23	1.04

L[1] Contiguous U.S. states (without Hawaii and Alaska)
Total land area of the USA = 916×10^6 ha

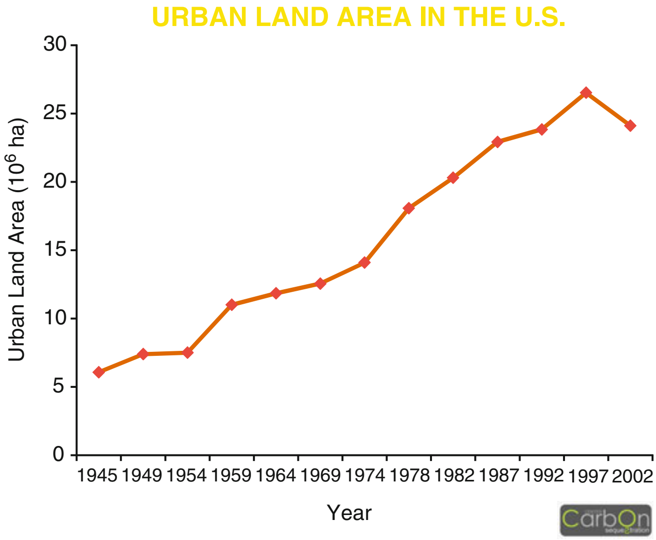

Fig. 19.1 Temporal changes in the urban land use in the U.S. (Redrawn from ERS 2010)

(Liu et al. 2010; Quan et al. 2007) and India (Dewan and Yamaguchi 2008). As a major force in land use conversion (Schneider and Woodcock 2008), urban sprawl adversely impacts ecosystem's integrity (Pino et al. 2000; Alberti et al. 2003), atmospheric abundance of GHGs (Shen et al. 2008), degradation of soil and water resources (Alphan 2003; Chen 2007), and exacerbation of global food insecurity (FAO 2009). The latter is partly driven by conversion of prime land to urban use (Morello et al. 2000), and severe competition for water (Haase 2009) and other limited resources. The resource depletion of distant lands which are converted to meet the deficit thus created is also an important factor. The impact of urbanization on the ecosystem carbon (C) pool and dynamics affects the global carbon cycle (CGG) and with the attendant impact on local, regional, and global climate.

19.3 Urbanization Impact on Carbon Pool and Dynamics

Urbanization disrupts the cycling of C and other elements, and also alters the hydrologic cycle and energy budget urbanization of natural and agroeco system, disrupt the C cycle (Svirejeva-Hopkins et al. 2004; Svirezhev 2002). There is a strong coupling of the C cycle with those of H_2O and other elements, which also strongly interact with the energy balance. Therefore, assessing the impact of urbanization on the GCC is a high priority to identify options to restore and enhance the ecosystem C pool and its functions (Steffen 2006), with drastic impact on environment and ecosystems. Although, the total area is relatively small, its impact on the GCC cannot be ignored.

Odum (1971, 1983) proposed that urban areas are not homogenous but comprised of two distinct components: (i) build up (P_1) and (ii) green areas or free space (P_2). The latter consists of open spaces covered by lawns, parks, sport grounds, recreational areas and forests, etc. These green areas (P_2) are linked to the GCC albeit in an altered/transformed state (Svirejeva-Hopkins and Schellnhuber 2006). Thus, GCC in urban ecosystems can be described in relation to these components, specifically, assessment of the relative proportion of the green areas within the urban centers, and its C pool and dynamics are important to evaluating their role in the GCC. Both quantity and quality of C pool in P_2 areas are important to GCC. Using this concept, Svirejeva-Hopkins and Schellnhuber (2006) developed a conceptual model to assess the role of urban centers in the GCC.

19.4 Greening of Urban Ecosystems and Researchable Priorities

The strategy is to increase the green city area, enhance its net primary productivity (NPP), increase both the living biomass and dead organic matter in these areas, and enhance the over all ecosystem C budget. This volume has specifically focused on enhancing the C pool in home lawns (Zirkle et al., Chap. 14), golf courses (Selhorst and Lal, Chap. 13), urban forests (Pouyat et al., Chap. 5), urban soils (Qian, Chap. 8) and urban agriculture (Beniston and Lal, Chap. 15), etc. The objective is to create a net positive C budget in the green city areas of the urban ecosystems. Some researchable priorities to realize this objective listed in Table 19.6 include those aimed at: (i) decreasing the hidden C cost (HCC), (ii) creating a positive net ecosystem C budget, (iii) enhancing depth distribution of soil C, (iv) increasing the below-ground C pool, (v) using integrated nutrient management (INM), (vi) choosing integrated pest management (IPM), (vii) enhancing net primary productivity (NPP), (viii) choosing adaptable species, (ix) increasing use efficiency of inputs, and (x) assessing co-benefits.

Developing measurement/modeling techniques to estimate the ecosystem C pool in green space of the urban ecosystems is a high priority. Credible estimates of the current pool and the technical potential of C sequestration are needed to identify policy interventions for sustainable development of these ecosystems which are rapidly expanding during the twenty-first century.

Urban ecosystems have a high C foot-print (Hillman and Ramaswami 2010). Gaseous emissions related to energy use and transport etc. can be off-set by sequestering C within the green area. Sustainable development of urban agriculture (UA) is important to the overall strategy of reducing the C-foot print of urban centers. Promoting sustainable UA would reduce the energy needs for food and fuel (food mileage related to long-distance freight trucking). Another researchable issue of great significance is the processing of waste/water and recycling of nutrients. Water and nutrients must be efficiently recycled to produce food (i.e., aquaponics) and biofuel feedstock (algae, cyanobacteria).

Table 19.6 Researchable priorities towards greening of urban lands and carbon sequestration in urban ecosystems

Urban land use	Research priorities
I Home lawns	• Decreasing hidden C costs • Enhancing use efficiency of inputs • Choosing adaptable species • Managing lawn mowing • Prioritizing biomes/regions • Integrating lawns with trees and shrubs • Reducing emissions of N_2O, CH_4, etc. • Enhancing depth-distribution of soil carbon • Making C pool in lawn independent of climate
II Recreational lands/turfs	• Minimizing the hidden C costs • Reducing inputs • Decreasing soil compaction • Improving aeration • Integrating trees and shrubs • Prolonging the duration of C sequestration • Establishing turfs on degraded lands • Increasing use efficiency of inputs
III Urban trees/forests	• Choosing adaptable species • Developing management guidelines • Enhancing below-ground C pool • Using IPM techniques • Evaluating co-benefits of trees • Managing micronutrients (Fe, Mn) • Enhancing ecosystem C pool
IV Urban agriculture	• Increasing agronomic productivity • Improving soil structure and tilth • Enhancing effective rooting depth • Creating balanced nutrients by INM • Reducing impacts of heavy metals (Pb) • Creating a favorable water regime • Developing appropriate seedbed • Using IPM • Creating a positive C budget

19.5 Linking Science with Policy

Moving ahead with implementation of researchable priorities outlined in the previous section necessitates linking of science with policy. The goal is to drive the policy by scientific principles rather than by perception. Policy instruments must be guided by the scientific principles of ecosystem management, especially those with a focus on recarbonization of the green areas within the urban centers. The strategy is to protect green ecosystems, restore degraded/depleted ecosystems, and accentuate co-benefits by integrated approach to resource management (Lyons 1997). Policy interventions, through communication with policy makers, are essential to recognize and strengthen inter-connectedness of these important resources. Policy interventions are needed to protect and restore these resources by enhancing scientific understanding of the structures, functions, and processes of green areas within the urban ecosystems.

While understanding the biophysical processes and the underlying scientific principles, the importance of the issues pertaining to the human dimensions cannot be over–emphasized: It is essential to create awareness among policy makers and the general public about the importance of these ecosystems. Since individual home/ property owners are members of the urban community, it is necessary to understand the decision making process at individual, community, urban center, watershed and regional level. Such a hierarchical linkage necessitates establishing and strengthening of the channels of communication between the scientific community on one side and general public and policy makers on the other.

19.6 Conclusions

Urban ecosystem constitute an important land use at local, national, regional and global levels. Urban centers are also major emitters of GHGs because of proportionately large consumption of electricity, diesel and other transport fuel, food and other consumables. Greening of urban ecosystems, therefore, implies an efficient use of natural resources on the one hand and sequestering C within the green space areas on the other. The latter consists of home and corporate lawns, sports and recreational grounds, urban trees/shrubs and forests, and UA. The ecosystem C pools of most of these green areas have been depleted because of drastic perturbations. Therefore, the strategy is to protect, restore and sequester C in the above and below-ground biomass, and in the soil C pool. Major researchable priorities of C sequestration in the home lawns and turfs include identification of strategies to minimize the HCC, increase use-efficiency of energy-based inputs, accentuate depth distribution of soil C, etc. Sustainable management of UA, while enhancing agronomic production, can also improve soil quality and increase C storage in soils. Principal soil-related constraints, which constitute researchable priorities, are identification of management strategies which improve soil structure and tilth, increase effective-rooting depth, create a favorable soil moisture regime, promote balanced application of plant nutrients, and minimize risks of contamination by heavy metals (i.e., Pb).

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