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Managing soil carbon stocks to enhance the resilience of urban ecosystems

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Abstract

Land use and land cover change (LULCC) by urbanization will likely replace agricultural expansion as dominant source of transformation of the terrestrial biosphere. Global-scale urbanization causes, in particular, carbon (C) losses from soil and vegetation, and those losses are projected to increase in the future as an area the size of South Africa may be converted to urban ecosystems in the coming decades. However, properly managed urban soils can off-set some of the losses by retaining C from the atmosphere as stabilized soil organic carbon (SOC) or soil inorganic carbon (SIC) such as mineral-associated organic matter (OM), black carbon (BC), and stable inorganic or carbonate minerals. Based on an up-date of previous reviews on urban soil C storage, SOC stocks of up to 810 Mg C ha^{-1} to 1.5-m depth (Serebryanye Prudy, Russia) and SIC stocks of up to 300 Mg C ha^{-1} to 2.5-m depth (Newcastle upon Tyne, U.K.) have been reported, but data on urban soil C storage are scanty. Aside contributing to climate change mitigation by C storage/regulation, protecting and increasing SOC stocks support critically important soil-derived ecosystem services, including nutrient cycling, primary production, water filtration and storage, erosion control, soil strength and stability, pollutant attenuation and regulation of biodiversity. Thus, the C capture function of urban soils within the constructed environment should be considered when planning for new or existing developments. Further, C-friendly soil and land-use management practices must be developed and implemented to enhance: (i) soil-derived ecosystem services in urban areas, and (ii) the resilience of urban ecosystems to climate change. A collective management approach for urban soil C is needed. The principal actors involved should be urban land users (e.g., urban dwellers, property owners, developers) as the immediate users and managers of soil C, local professionals, local government and NGOs.

Key words: Soil inorganic carbon; Soil organic carbon; Soil-derived ecosystem services; Climate change adaptation and mitigation; Urban environmental quality

Introduction

Many ecosystems will be at risk when temperature increases by more than 2 degrees Celsius relative to pre-industrial times (Randalls, 2010). To keep global warming below this temperature increase, it is crucial to limit the input of fossil fuel carbon (C) into the atmosphere (Hansen et al., 2013). However, C storage in the biosphere, including the soil, via reforestation and improved agricultural and forestry practices can potentially offset some of the anthropogenic C emissions (Scharlemann et al., 2014). Aside agricultural expansion and deforestation, land use and land cover change (LULCC) by urbanization is one of the key drivers of global change as it causes C losses from the terrestrial biosphere. Currently, the urban land area occupies an estimated 0.5% of the total global land area (Schneider et al., 2009). However, the global urban extent is uncertain as, for example, the global area under settlements may be much larger than previously thought based on the evaluation of satellite data in unprecedented spatial resolution of ~12 m by the Global Urban Footprint project (Figure 1). Further, assessments of urban land cover change are hampered as no global data set on urban expansion exist (Fragkias and Seto, 2012). A recent meta-analysis indicated an increase of urban land area by almost 60,000 km^2 between 1970 and 2000, and urban land cover may drastically increase by 1.2 million km^2 by 2030 (Seto et al., 2011; 2012). Between 2000 and 2010, for example, compactness decreased and sprawl increased for large urbanized areas in the U.S. (Hamidi and Ewing, 2014). Such urban expansion occurs commonly at the expense of agricultural land as many cities were founded in agricultural areas on coastal plains and in river valleys (Hooke et al., 2012). Further, urban expansion may also become a driver of deforestation in tropical regions (Seto et al., 2012). Thus, conversion of land to urban uses potentially results in the loss of terrestrial C including soil organic carbon (SOC) (e.g., Xiong et al., 2014). Up to 0.05 Pg C (1 Pg = 1 petagram = 10^{15} g) may be lost annually from vegetation biomass in the pan-tropics by deforestation and forest degradation for urban expansion (Seto et al., 2012). This loss is not negligible compared to the global carbon dioxide (CO_2) emissions from other land use changes and deforestation amounting on average to 0.9 Pg C annually during 2003-2012 (Le Quéré et al., 2013). Further, with cities contributing to more than 70 percent of fossil fuel related CO_2 emissions (EIA, 2013), it is evident that when aggregated globally, local processes within urban areas have the potential to affect the Earth system (Grimm et al., 2008). However, the biogenic exchange of C in urban soils and vegetation is a fundamental, yet insufficiently understood, C flow occurring differentially within and across different urban areas (Hutyra et al., 2014).

Land use and land cover change effects on urban soil carbon

Urban LULCC can directly alter above- and belowground biomass C, and soil C that is comprised of soil inorganic carbon (SIC) and soil organic carbon (SOC). To effectively offset the C emissions for climate change mitigation, C sequestration implies that C must remain stored not just for 100 yr, but probably for more than 10,000 yr due to the long atmospheric residence time of some of the emitted C (Hansen et al., 2013; Mackey et al., 2013). This long-term terrestrial C storage occurs, in particular, in soils. Under the environmental conditions of most soils, bicarbonate and carbonate ionic forms, as well as carbonated salts in the solid phase are among the predominant stable forms of C (Macías and Arbestain, 2010). Carbon dating of SIC (i.e., pedogenic carbonates) indicates long C residence times of > 30,000 yr (Renforth et al., 2009). Similarly, radiocarbon ages of 1,000 yr to >10,000 yr have been reported for SOC, especially in sub-soil layers, and SOC turnover times increase with increase in soil depth (Rumpel and Kögel-Knabner, 2011).

Not the intrinsic properties of SOC itself but rather physicochemical and biological influences allow SOC to persist for long periods of time (Schmidt et al., 2011). For example, Courtier-Murias et al. (2013) emphasized that the main mechanism by which soil C inputs are stabilized and SOC accrues is the adsorption of microbial biomass and microbial by-products on mineral surfaces rather than the physical and chemical protection of undecayed or partially degraded organic structures. Organic amendments may increase more than previously thought the microbial populations of the soil, which live, thrive, and die in close association with the mineral surfaces. The joint physical-chemical mechanism of SOC stabilization may be enhanced by the addition of organic matter (OM) relatively richer in compounds with molecular structures and/or assemblies more resistant to decomposition, e.g., condensed aromatic C added with combustion-derived compounds, and aliphatic C added with inputs of cutin and suberin (Courtier-Murias et al., 2013; Marschner et al., 2008; Lorenz et al., 2007). The association of SOC with minerals and, in particular, organo-mineral clusters on clay-sized particles with rough surfaces may be the most important factor in SOC stabilization (Vogel et al., 2014). Stabilization may increase with increase in soil depth, irrespective of vegetation, soil type, and land use (Schrumpf et al., 2013). However, the reasons for the very long SOC turnover times are not completely understood, and little is also known about how carbonates form in anthropogenic environments (Renforth et al., 2011; Schmidt et al., 2011). Thus, there is a need for better soil C science to mitigate losses and enhance soil C stocks (Scharlemann et al., 2014).

Previously, C dynamic and storage in urban vegetation has received some attention (e.g., Muñoz-Vallés et al., 2013). However, soil is often the largest contributor to urban C storage. For example, urban soils of China and the conterminous United States stored about 56% and 64% of total urban C, respectively (Churkina et al., 2010; Zhao et al., 2013). Only a few reviews/studies on urban SIC and SOC dynamic and stock have been published (e.g., Lorenz and Lal 2009; Luo et al. 2012; Pouyat et al. 2006; Rawlins et al. 2008; Tong and Dong 2007).

Urban soil C data are, especially, scanty for major regions and countries (e.g., Canada, India, Near and Middle East, Central and South America, Oceania, Southeast Asia). Most importantly, data are missing for cities and urban regions in Africa although the strongest urban expansion is projected to occur on this continent in the coming decades (Seto et al. 2012). Thus, the importance of urban soil C stocks continues to be overlooked (e.g., Stockmann et al., 2013). However, data on urban soil C are needed for an integrated understanding of the processes of urbanization and the impacts of urban areas on C flows (Romero-Lankao et al. 2014).

Ecosystem services of urban soils

The C storage/regulation is among the provisioning services of urban soils. The ecosystem services provided by soils are the result of inherent and manageable soil properties (Morel et al. 2014). Urban soil ecosystem services can be distinguished based on the Millennium Ecosystem Assessment typology. Supporting services of urban soils (e.g., nutrient cycling, primary production) are regarded less important compared to those of non-urban soils (Rawlins et al. 2013). However, increasing demand for food produced by urban agriculture may substantially enhance primary production and associated nutrient cycling in urban soils. Among the provisioning ecosystem services, the importance of urban soils for food production and C storage/regulation are, therefore, also increasing although to a lower extent than those of non-urban soils. Otherwise, flood regulation by urban soils is regarded as being of low importance on a regional scale but with a large potential for enhancement of local water storage, reduction in surface runoff and flood control (Rawlins et al. 2013). Other regulating services of urban soils are pollution attenuation and regulation of biodiversity (Morel et al. 2014). Important cultural services of vegetated urban green space which rely on soils for their biogeochemical cycling include recreation/tourism and cultural heritage. Urban soils fulfil also a range of functions not addressed by the Millennium Ecosystem Assessment (MEA) such as carrying structures and piped utilities (Rawlins et al. 2013).

The importance of soil carbon for urban ecosystem services and urban ecosystem resilience

The urban soil C, i.e., both SIC and SOC contribute to the provisioning ecosystem service C storage/regulation (Rawlins et al. 2013). Further, SIC also supports other urban soil functions and derived ecosystem services. For example, in many urban soils dissolution of soil carbonates is the dominant buffering mechanism which limits soil acidification (Ming 2006). Thus, by contributing to optimal soil pH for plant growth, soil carbonates may also support net primary production (NPP) in urban ecosystems. Further, surfaces of calcite [$CaCO_3$] are reactive and various ions may adsorb or interact with them (Ming 2006). Ions essential to plants may be adsorbed and this adversely affects their availability for plant uptake with repercussions on some ecosystem services (e.g., NPP, food production). Otherwise, urban soil contaminants (e.g., Ba^{2+}, Cd^{2+}, Pb^{2+}) may be precipitated by soil carbonates and this process contributes to the urban soil ecosystem service pollutant attenuation (Morel et al. 2014).

The SOC plays a key role for supporting, regulating, provisioning and cultural soil ecosystem services (Banwart et al. 2014). The benefits of SOC for soil functions and ecosystems services are well known and related to the fundamental role of SOC in the function and fertility of terrestrial ecosystems (Janzen, 2006). The SOC is a key indicator of soil quality as it affects essential biological, chemical and physical soil functions such as (i) plant nutrient cycling, (ii) pesticide, pollutant and water retention, (iii) energy supply for microorganisms and, (iv) soil structure maintenance (Karlen et al. 1997; Mueller et al. 2010). The ability of ecosystems to provide soil ecosystem services depends on soil quality and, thus, on SOC (Lal 2010a). Specifically, the cycling of SOC can be managed to amplify ecosystem services, improve water quality, increase biodiversity and NPP (Lal 2010b). To sustain the manifold functions performed by ecosystems maximizing SOC 'stocks' is less critical than maintaining SOC 'flows' (Janzen 2015). Thus, ecosystem services are improved by an increase in the SOC pool (Franzluebbers 2010). Specifically, urban SOC is important for nutrient cycling, primary production, food production, erosion control, water storage, pollution attenuation, biodiversity regulation, and recreational and cultural services. Urban soil C contributes to urban ecosystem resilience or 'the ability of a system to absorb disturbance and still retain its basic function and structure' (Walker and Salt, 2006). For example, SIC buffers urban soils and derived ecosystem services against detrimental effects of inputs of acidifying compounds. Further, SOC supports urban agricultural production and, thus, can contribute to long-term food security during areas of energy scarcity (Barthel and Isendhal, 2013). By its effect on biodiversity regulation, SOC contributes to the fundamental role of biodiversity in building resilience in urban systems (Jansson 2013). However, an urban ecosystem management based on ecological resilience should also emphasize the heterogeneity in SOC stocks as this variability also contributes to urban ecosystem resilience (Holling 1973).

There are strong interactions between SOC sequestration and soil, water and air quality (Smith et al., 2013). As belowground communities strongly control SOC sequestration, microbial diversity and functional capabilities should be used to guide and monitor urban soil management and restoration (Fierer et al., 2013). The long lasting legacy of urban investment decisions, growth of urban activities that produce CO_2 emissions, and increasing concentration of urban population drives urgency for urban climate change policy (Marcotullio et al., 2013), and for soil C-friendly urban soil and land-use management (Seto et al., 2012).

Protecting inorganic carbon stocks of urban soils

The SIC comprises both primary carbonates inherited from the parent material or deposited as dusts, and secondary carbonates which form by precipitation of carbonate ions derived from root, microbial and faunal respiration, and calcium (Ca) and magnesium (Mg) ions from weathering (Lal and Kimble, 2000). However, very little is known about SIC as studies on soil C have often focused on SOC. Thus, the global SIC stock may be much higher than previously reported as national and regional databases seldom provide data (Eswaran et al., 2000; Rawlins et al., 2011). Especially in arid and semiarid regions, the SIC stock may be up to ten times larger than the SOC stock (Eswaran et al., 2000). Studies of carbonates in ancient soils indicate their stable nature (residence times of up to 2.6 Ga; Watanabe et al., 2004).

Carbonate-bearing soil parent materials commonly occur in urban soils (Lehmann and Stahr, 2007). In addition, demolition waste (particularly cement and concrete) may contribute to urban SIC storage (Washbourne et al., 2012). For example, the $CaCO_3$ contents of technogenic materials in urban soils range between 0% and >10% (Meuser, 1992). Thus, urban soils potentially contain SIC aside SOC. Especially, the coarse fraction (> 2mm) may substantially contain SIC in the form of demolition waste, limestone and chalk fragments (Rawlins et al., 2011). In the following section, studies on urban SIC stocks from various global regions are briefly discussed. Some SIC stocks are calculated from reported carbonate stocks.

Along the urban Interstate 64 in Louisville, KY, USA, SIC storage to 15-cm depth was compared for plots with high-density cover of the shrub Amur honeysuckle (*Lonicera maackii*) growing on disturbed and on undisturbed soil, and low-density honeysuckle cover on undisturbed soil (Table 1; Trammell and Carreiro, 2012). Soils with undisturbed soil horizons tended to have higher SIC stocks (0.9 Mg C ha^{-1}) than those that were highly disturbed during highway construction as the sub-soil was exposed at the surface. Further, high-density honeysuckle cover plots tended to have lower SIC stocks (1.7 Mg C ha^{-1}) to 15-cm depth than their low-density counterparts (Trammell and Carreiro, 2012).

The average SIC stock to 20-cm depth among five major cities in the Jiangsu Province, China, was 6.4 Mg C ha^{-1}, and ranged between 0 and 49.5 Mg C ha^{-1} (Table 1; Xu and Liu, 2013).. On average, 1.33 times and 1.52 times more SIC were stored in urban soils of the five cities than in suburban and countryside soils, respectively.. The accumulation of SIC in central urban areas was obviously caused by high carbonate and bicarbonate inputs with long-lasting groundwater deposition (Xu and Liu, 2013).

The accumulation of SIC in urban soils of Shanghai has probably also been caused by surface deposition of carbonate-rich groundwater (Xu et al., 2012). The average SIC stock to 20-cm depth for Shanghai was 11.2 Mg C ha^{-1}, ranging between 1.3 and 30.7 Mg C ha^{-1} (Table 1). The average SIC stock was 1.53 times higher than that of suburban soils while countryside soils stored 1.16 times more SIC than urban soils.. At 160-180 cm depth, average SIC stocks in Shanghai were 12.4 Mg C ha^{-1}, and varied strongly between 0.9 and 21.3 Mg C ha^{-1} (Table 1). Urban soils stored 1.13 times more SIC at 160-180 cm depth than suburban soils, and countryside soils stored 1.27 times more SIC than urban soils to the same depth.. In conclusion, the SIC stocks in Shanghai increased in the central urban area with urban land use duration and urban ecosystem evolving. However, SIC stocks of urban areas in Shanghai at both depths were lower than SOC stocks (Figure 2; Xu et al., 2012).

Soil profile carbonate stocks were determined for 11 land uses at 33 sites in Stuttgart, Germany (Stahr et al., 2003). The SIC stocks to 30-cm depth within the city ranged between 12 Mg C ha^{-1} at railway areas to 82 Mg C ha^{-1} at vineyards (Table 1). In comparison, rural forest soils contained no carbonates while rural agricultural soils stored 7 Mg SIC ha^{-1}. At 30-100 cm depth, urban SIC stocks ranged between 67 Mg C ha^{-1} at military barracks and 266 Mg C ha^{-1} at the old city center. Again, rural forests soils were carbonate-free at 30-100 cm depth while rural agricultural soils contained 2 Mg SIC ha^{-1}. Soil addition of calcareous building rubble and limestone gravel together with the carbonate-rich soil parent material contributed to increased profile SIC stocks. Decalcification caused SIC losses from surface soils and contributed to SIC accumulation in sub-soil layers. Further, urban soils with high proportion of coarse fraction contained less SIC (Stahr et al., 2003). However, SOC stocks in soil profiles in Stuttgart were frequently higher than the SIC stocks (Lorenz et al., 2003).

In Newcastle upon Tyne, UK, the addition of Ca-rich demolition waste to a brownfield site resulted in highly variable SIC stocks (Renforth et al., 2009). The soil had accumulated on average 25 Mg SIC ha^{-1} y^{-1} to 2.5-m depth and stored 300 Mg C ha^{-1} as $CaCO_3$ which was three times more than the SOC stock. About 97% of the SIC has been sequestered ultimately from the atmosphere, i.e., 57% of the carbonate C was of organic origin from photosynthesis and 40% derived from hydroxylation (Renforth et al., 2009).

The SIC stocks to 30-cm depth under lawns in Fort Collins, CO, USA, ranged from 1.3 Mg C ha^{-1} in 0-15 cm to 2.0 Mg C ha^{-1} in 15-30 cm depth (Table 1; Kaye et al., 2005). In comparison, SOC stocks at 0-15 cm and 15-30 cm depths were 47.6 Mg C ha^{-1} and 21.8 Mg C ha^{-1}, respectively. The SIC stocks were not different from those in adjacent native and agricultural ecosystems (Kaye et al., 2005). In contrast, urban soils of xeric and mesic yards and nonresidential land uses in Phoenix, AZ, USA, contained more SIC at 0-10 cm and 10-30 cm depths (i.e., 0.45-0.62 and 0.98-1.04 Mg C ha^{-1}, respectively) than desert soils (Table 1; Kaye et al., 2008).. Irrigation with water saturated with $CaCO_3$ probably contributed to SIC accumulation. The SIC stock to 60-cm depth of green spaces in two residential blocks in Chicago, IL, USA, was on average 43 Mg C ha^{-1} which was four times less than the amount stored as SOC (Table 1; Jo and McPherson, 1995).

In conclusion, knowledge about SIC dynamic and stock in urban soils is scanty making it difficult to recommend practices for protecting the existing urban SIC stocks. However, priority should be given to reducing the loss of urban soil containing SIC to rural areas as thermodynamically very stable C sequestered as carbonates may be lost from urban ecosystems. Minimizing physical removal of soil must include measures to reduce soil erosion losses, in particular, during construction activities when urban soil is left uncovered. Both excavated carbonate-containing soil and coarse fraction should be used in situ for landfilling and landscaping activities within the same urban area. Special attention should be given to urban soils heavily affected in the past by intense demolition activities following military conflicts (e.g., World War II) and economic decline (e.g., Rust Belt cities in the USA) as those soils may contain high SIC stocks as a product of carbonation of deposited concrete- and cement-derived material. Also, urban soils of arid and semiarid regions should be carefully managed for SIC preservation as they may naturally contain high SIC stocks.

Protecting organic carbon stocks of urban soils

Previous reviews on urban SOC stocks were based on a limited number of studies and sometimes on assumptions with regard to SOC depth distribution and storage under impervious surfaces (e.g., Pouyat et al., 2006). Nevertheless, it became obvious that SOC stocks within urban ecosystems are highly variable. For example, SOC stocks to 30-cm depth in Stuttgart, Germany, were as low as 7 Mg C ha^{-1} and as high as 232 Mg C ha^{-1} (Stahr et al., 2003). Further, in 0-100 cm depth, SOC stocks in Stuttgart ranged between 15 Mg C ha^{-1} and 285 Mg C ha^{-1}, and this range was comparable to that of soils in terrestrial biomes but on a much smaller spatial scale (Lorenz and Lal, 2009). In the following section, recently published data on SOC stocks for urban areas in diverse global regions are discussed. Also presented are data from studies reporting only total urban soil C stocks for soil pH of < 7.

Australia

The SOC stocks of lawns and wood chip mulched gardens in Victoria were statistically similar at both 0-10 cm and 10-25 cm depths, respectively (Table 2; Livesley et al., 2010). However, long-term (50-100 y) OM inputs with mulch resulted in an increase in SOC concentrations and caused a decrease in soil bulk density (Livesley et al., 2010).

Canada

In a low-density neighborhood in Vancouver, B.C., urban soils under lawns covered by some trees stored on average 96.5 Mg SOC ha^{-1} to 25-cm depth (Table 2; Kellett et al., 2013). Across the entire neighborhood, urban soils stored on average 34.5 Mg SOC ha^{-1} to this depth. However, the estimate is uncertain as it was based on the assumption that urban soils under impervious surfaces contained no SOC and that the pervious surface fraction in the entire neighborhood was 35.7% (Kellett et al., 2013).

China

Zhao et al. (2013) estimated that SOC stocks to 100-cm depth for urban green spaces ranged between 47.5 Mg C ha^{-1} in Xinjiang, Northwest China, and 184.6 Mg C ha^{-1} in Hubei, Central-south China (Table 2). The highest SOC stocks and the highest proportion of total urban C stored as SOC (i.e., 70%) were reported for soils under cold climate in the Northeast region. In contrast, urban green space soils in East China stored the lowest proportion of total urban C as SOC (i.e., 50%). However, those estimates were based on the assumption that SOC density below impervious surfaces is stable once the soil is covered. For example, the stable SOC stocks to 100-cm depth under impervious cover were estimated to range between 46.0 Mg C ha^{-1} in Xinjiang and 234.0 Mg C ha^{-1} in Heilongjiang, Northeast China (Zhao et al., 2013).

Hao et al. (2013) reported that the SOC stock to 30-cm depth for Tianjin Binhai New Area was 174.4 Mg C ha^{-1} (Table 2). In comparison, SOC stock to 30-cm depth of urban green areas was 91.7 Mg C ha^{-1} . However, estimates for the entire area were based on the assumption that between 5.0 and 41.0 Mg SOC ha^{-1} were stored to 30-cm depth under paved surfaces and that no changes in SOC stocks occurred after paving. Continuous manure and fertilizer applications improved urban green space soils but overall urbanization at Tianjin Binhai New Area has resulted in high SOC losses (Hao et al., 2013).

The SOC stocks to 20-cm depth of green space soils in Nanjing City were 45.2 Mg C ha^{-1} and, thus, higher than those of impervious-covered soils (23.5 Mg C ha^{-1}; Wei et al., 2014). It was hypothesized that reduced C input (e.g., with plant litter, manure) and the removal of the original SOC-rich arable soil layer resulted in lower SOC stocks of impervious-covered soils (Wei et al., 2014).

In Chongqing Municipality, SOC stocks to 20-cm depth of green spaces were on average 26.1 Mg C ha^{-1} (Liu et al., 2013). The SOC stocks exhibited a high spatial variability, i.e., varying between different types of green land, different slopes, and different urbanized periods. For example, SOC stocks to 20-cm depth of urban parks and gardens were 36.3 Mg C ha^{-1}, and ranged from 12.8 Mg C ha^{-1} at a newly established park to 156.5 Mg C ha^{-1} at Chongqing Zoo (Table 2).. Low SOC stocks in scattered transport and street green land were the result of reduced fertilization and watering. Intense human activities exacerbated soil erosion and caused a loss of SOC and soil fertility.. To accumulate SOC in soils of Chongqing Municipality, Liu et al. (2013) recommended the integration of nutrient management and waste recycling.

In Kaifeng city, average SOC stocks to 10-cm depth of green spaces were 24.5 Mg C ha^{-1} but varied among urban districts (Sun et al., 2010). Specifically, SOC stocks were 19.1, 22.1, 25.2, 31.2 and 33.7 Mg C ha^{-1} for residential/administrative, cultural/educational, traffic, industrial, and recreational districts, respectively (Table 2).. In 0-100 cm depth, average SOC stocks were 99.7 Mg C ha^{-1}, and stocks were the highest in the cultural/educational and traffic districts (110.2 Mg C ha^{-1}) and decreased in the order industrial, recreational and residential/administrative districts (105.7, 88.4 and 69.9 Mg C ha^{-1}, respectively; Table 2).. The depth distribution of SOC was strongly altered by human activities.. Sun et al. (2010) concluded that frequent fertilization, watering and scarification enhanced plant growth in urban green spaces and this, together with OM inputs, contributed to SOC accumulation in urban relative to suburban soils of Kaifeng city.

The average SOC stocks at 0-20 and 160-180 cm depth in Shanghai were 39.3 and 15.5 Mg C ha^{-1}, respectively, but highly variable (Table 2; Xu et al., 2012). The observed SOC accumulation at both depths in the central urban area may have resulted from the practice of ecological city construction that was associated with a strong increase in green coverage in Shanghai (Xu et al., 2012).

The SOC stocks to 15-cm depth of turf grass sites in Hong Kong varied between 12.6 and 48.9 Mg C ha^{-1}, and depended on land use and time since site establishment (Table 2; Kong et al., 2014).. The SOC stocks of turf grass lawns in Hong Kong remained relatively stable over five decades but turf management practices may ultimately determine whether the lawn sites are net sources or sinks of CO_2 (Kong et al., 2014).

The SOC stocks to 30-cm depth for rural village landscapes in China varied depending on land use and region (Jiao et al., 2010). For example, stocks to 30-cm depth were as low as 5.2 Mg C ha^{-1} for lands with mine and fill use in the subtropical hilly region, and as high as 48.5 Mg C ha^{-1} for ornamental gardens in schoolyards and lands along a highway section in the North China Plain (Table 2).. The human alteration of soils and vegetation may have had stronger influence on SOC stocks than macroclimate. Specifically, anthropogenic removal of topsoil and a sparse vegetation cover resulted in low SOC stocks in fallow land and mined areas.. The SOC stocks of sealed soils to 30-cm depth were on average 22.9 Mg C ha^{-1}, ranging from 16.7 to 31.6 Mg C ha^{-1}. In summary, the SOC stocks to 30-cm depth in and around built structures of village landscapes formed a substantial component of regional SOC stocks. Thus, human residence and not just agricultural practice is a regionally important control on SOC stocks across rural village landscapes in China (Jiao et al., 2010).

New Zealand

Under the putting greens of golf courses in Palmerston North, SOC stocks to 25-cm depth increased by 28 Mg C ha^{-1} over 40 years (Huh et al., 2008). This increase was higher than that in extensive pasture systems or forests.. Transformation of C into more resistant forms by biochemical alteration reduced the bioavailability and was identified as major mechanism for SOC sequestration in putting greens at Palmerston North (Huh et al., 2008).

Russia

Urban soils in the industrial city of Revda, Middle Ural, were studied by Meshcheryakov et al. (2005). Soils of city lawns and old vegetable gardens stored 119.2 and 137.8 Mg SOC ha^{-1} to 25-cm depth, respectively (Table 2; calculated from humus stock assuming %humus = 1.72 x %C_{org}). The SOC stocks were higherthan those of the technogenic desert planted with poplar trees and those of park soils characterized by a forest litter layer (Meshcheryakov et al., 2005).

The SOC stocks of functional zones in Moscow and Serebryanye Prudy were estimated by Vasenev et al. (2013) under the assumption that bulk density did not differ among soils. In Moscow, the SOC stocks to 10-cm depth were as low as 28.1 Mg C ha^{-1} and as high as 70.7 Mg C ha^{-1} at residential and recreational zones [1], respectively (Table 2). In comparison, SOC stocks at 10-150 cm depth were as low as 75.6 Mg C ha^{-1} and as high as 818.8 Mg C ha^{-1} for residential and recreational, zones, respectively. Differences in anthropogenic transformations contributed to the high variability in SOC stocks. In contrast, soils of the town Serebryanye Prudy were less disturbed. Specifically, SOC stocks at 0-10 and 10-150 cm depths for industrial, residential and recreational zones were 68.1, 49.8 and 68.6 Mg C ha^{-1}, and 850.0, 898.8 and 1077.5 Mg C ha^{-1}, respectively. The average SOC stocks to 150-cm depth in Moscow and Serebryanye Prudy were 128 and 810 Mg C ha^{-1}, respectively. About 25% and 5% of the total SOC stocks at 0-150 cm depth in Moscow and Serebryanye Prudy were stored to 10-cm depth, respectively (Figure 3). In comparison, rural soils near Moscow and Serebryanye Prudy had stored less SOC (Vasenev et al., 2013).

United Kingdom

The SOC stock to 21-cm depth of green spaces in Leicester was on average 99 Mg C ha^{-1} (Edmondson et al., 2014). The median SOC stocks under non-domestic herbaceous vegetation and garden shrubs and trees were 86 and 135 Mg C ha^{-1}, respectively (Table 2). In comparison, SOC stocks to 21-cm depth were the lowest in regional arable land. The soil types did not affect urban SOC stocks but garden management practices such as the addition of peat, composts, and mulches, and cultivation of trees and shrubs contributed to greater SOC stocks in Leicester to 21-cm depth (Edmondson et al., 2014). Previously, Edmondson et al. (2012) reported that urban soils in Leicester stored more SOC to 100-cm depth than regional arable soils. Specifically, SOC stocks for non-residential and residential land were 161 and 199 Mg C ha^{-1} (Table 2).. About 42% of the urban SOC stock to 100-cm depth in Leicester was stored within the top 20-cm. However, SOC stocks were not affected by surface cover (i.e., vegetated or capped).. Edmondson et al. (2012) concluded that the contribution of urban ecosystems to national organic C inventories cannot be neglected.

USA

The SOC stocks to 10-cm depth of residential lawns in Richmond, VA, were on average 43.4 Mg C ha^{-1} but similar among occupied and vacant lots (Table 2; Gough and Elliott, 2012).. It was hypothesized that SOC stocks of abandoned lawns may approach those of forests several decades following conversion to residential land uses (Gough and Elliott, 2012).

The soil C storage of residential lawns and up to 51 y of home age were studied by Huyler et al. (2014a) in Auburn, AL. The SOC stocks at 0-15, 15-30 and 30-50 cm depth were 23.7, 9.8 and 5.1 Mg C ha^{-1}, respectively, and differences among sites were not related to yard maintenance practices (fertilization, irrigation, and bagging or mulching lawn clippings) or soil texture (Table 2). In a subsequent study, SOC stocks in Auburn were studied in yards containing trees with home ages ranging from 3 to 87 y (Huyler et al., 2014b). The mean SOC stocks at 0-15, 15-30 and 30-50 cm depths were 32.5, 10.3 and 7.4 Mg C ha^{-1}, respectively (Table 2). Trees in turf grass yards may have had a stabilizing effect on SOC stocks below 15-cm depth but minimal influence above this depth. However, tree aboveground biomass was a poor overall representative of SOC stocks (Huyler et al., 2014b).

The SOC stocks to 10-cm depth of urban mesic yards in Phoenix, AZ, were higher than those of urban xeric yards (11.0 vs. 5.0 Mg C ha^{-1}; Table 2; Kaye et al., 2008). At 10-30 cm depth, SOC stocks of xeric and mesic yards, and non-residential land uses ranged between 5.3 and 7.3 Mg C ha^{-1}. To 30-cm depth, mesic residential yards stored more SOC than xeric residential yards (Kaye et al., 2008).

The SOC stocks to 90-cm depth of urban lawns and forests near Apalachicola, FL, were not significantly different (i.e., 107 and 159 Mg C ha^{-1}), and also similar to those of natural pine forests and pine plantations (Table 2; Nagy et al., 2014). In comparison, forested wetlands contained much more SOC (i.e., 633 Mg C ha^{-1})..

The SOC stocks to 10-cm depth in Boston, MA, tended to decrease from forest to residential to other developed land uses (Table 2; Raciti et al., 2012b). Further, the average SOC stocks to 10-cm depth for other developed land uses (36 Mg C ha^{-1}) were lower than those for residential (40 Mg C ha^{-1}) and forest land uses (42 Mg C ha^{-1}), respectively. Higher stocks in each land use class were for the high population category and the lower stocks for the low population category.. However, population density correlated only weakly with SOC stocks. The main drivers for marginally higher urban SOC stocks to 10-cm depth in Boston may have been inputs of N (Raciti et al., 2012b).

Raciti et al. (2011) determined the SOC stocks to 100-cm depth for residential home lawns in Baltimore, MD, on similar soil types but different previous land use and age. Average SOC stocks under lawns were higher than those of forested reference sites (69.5 *vs*. 54.4 Mg C ha^{-1}). The residential lawn SOC stocks increased with increase in housing age but housing density was not a predictor for SOC stocks. Lawn soils in residential areas on former agricultural land in Baltimore had a significant capacity to sequester C to 100-cm depth (Raciti et al., 2011).

The average SOC stock to 15-cm depth under impervious cover among four neighborhoods in New York City, NY, was 22.9 Mg C ha^{-1} (Raciti et al., 2012a). In comparison, 56.7 Mg SOC ha^{-1} was stored to the same depth in open urban areas (Table 2).. Thus, SOC stocks under impervious urban surfaces cannot be neglected as a component of urban SOC storage. However, the fate of SOC lost or removed from open areas during construction of impervious surfaces must also be assessed (Raciti et al., 2012a).

The average SOC stock to 15-cm depth of residential urban forests in Bloomington, IN, was 47.2 Mg C ha^{-1} but varied strongly (Table 2; Schmitt-Harsh et al., 2013). This variation may probably be caused by the "legacy effect" (i.e., the influence of development age on C stock accumulation; Lewis et al. 2014). The average C density in soils to 15-cm depth was 1.9 times higher than those of trees highlighting the importance of soil protection for the urban ecosystem C balance (Schmitt-Harsh et al., 2013).

The SOC stocks to 15-cm depth of three urban forest soils adjacent to urban interstates in Louisville, KY, ranged between 30.3 and 80.5 Mg C ha^{-1} (Table 2, Trammell et al., 2011). At sites where imported fill was deposited post-construction the median SOC stocks were higher compared to sites where surface soil horizons were removed and sub-soil exposed at the surface .

The area-weighted SOC stock to 100-cm depth of urban soils in Baltimore, MD, was 71.1 Mg C ha^{-1} ranging between 105 and 121 Mg C ha^{-1} for remnant forests and turf grass sites, respectively (Table 2; Pouyat et al., 2009). However, across the city it was assumed that urban soils beneath impervious and unsealed surfaces stored 33.0 and 110.0 Mg C ha^{-1} to 100-cm depth, respectively.. Further, residential turf grass soils stored more SOC to 100-cm depth than rural forest soils (110 vs. 67 Mg C ha^{-1}).. Thus, residential turf grass soils in Baltimore have the capacity to accumulate large amounts of SOC (Pouyat et al., 2009).

The SOC stocks to 15-cm depth of home lawn turf grass soils of sixteen cities located in eight climatic regions across the conterminous U. S. were studied by Selhorst and Lal (2012). The average SOC stock was 59.8 Mg C ha^{-1} ranging from 34.9 Mg C ha^{-1} in Atlanta, GA, to 125.5 Mg C ha^{-1} in Minneapolis, MN (Table 2). The SOC stocks were the highest in regions of low mean annual temperature, moderate mean annual precipitation, high soil N concentration, and moderate bulk density. Thus, both climatic factors and soil properties affected SOC stocks in home lawn turf grass soils to 15-cm depth (Selhorst and Lal, 2012).

The SOC stocks to 20-cm depth of ornamental lawns established from seed on existing soil and those of athletic fields constructed from imported turf grass sods in Irvine, CA, were 12 and 35 Mg C ha^{-1}, respectively (Table 2; Townsend-Small and Czimczik, 2010).. Turf grass soils managed conservatively may contain high SOC stocks (Townsend-Small and Czimczik, 2010).

In conclusion, recently published data are in accord with previous reviews indicating that SOC stocks of urban ecosystems vary widely. However, major knowledge gaps still exist as data on SOC stocks are scanty for urban areas in Africa, Asia (except China and Japan), Latin America and Oceania. Further, SOC stocks for urban soil profiles are only sporadically reported as are those under impervious cover. Urban SOC stocks may be particularly high where urban LULCC occurred on naturally SOC-rich soils (e.g., organic soils, peatlands, permafrost) and where SOC stocks are enhanced through management practices such as addition/burial of organic C (e.g., manure, plant litter, construction debris), fertilization and irrigation. However, similar to SIC the SOC dynamics in urban soils are less well known. From studies in non-urban soils it can be deduced that protecting existing urban SOC stocks entails reducing physical soil disturbance during construction activities that may result in SOC loss to rural areas, the atmosphere and aquatic systems, and reducing decomposition losses during soil handling. Further, excavated urban soil should be used as fill material within the same urban area. The disturbance of vegetation cover on urban soils should be reduced to maintain existing SOC stocks. Thus, protecting existing urban SOC stocks should be focus of urban land use and planning as SOC has many environmental benefits including enhancing both soil-derived ecosystem services and the resilience of urban ecosystems discussed previously.

Enhancing urban soil carbon stocks

Enhancing urban soil C stocks can contribute to climate change mitigation by storing C for millennia. SOC sequestration, in particular, can also enhance soil, water and air quality in urban ecosystems. If designed and managed correctly, urban soils can retain C captured from the atmosphere as accumulated SOC or stable, inorganic, carbonate minerals (Renforth et al., 2011). This soil function should be considered for improving urban environmental quality when planning for new or existing developments. However, urban planning systems and, in particular, green space management practices are generally not aimed at enhancing soil C storage (Edmondson et al., 2014). Thus, awareness among stakeholders must be raised with regard to the positive effects of urban soil C, in particular, on soil-derived ecosystem services (Haase et al. 2014). Some examples of practices for enhancing SIC and SOC stocks in urban ecosystems will be given in the following section.

Enhancing inorganic carbon stocks of urban soils

A little is known about carbonate formation in urban soils (Renforth et al., 2011). Thus, recommendations of soil and land use management practices for enhancing urban SIC stocks discussed below are preliminary and need to be up-dated. Urban soils often contain Ca-carbonate minerals and some of which may be inherited from the soil parent material. The most common rock-forming carbonates are calcite ($CaCO_3$) and dolomite [$CaMg(CO_3)_2$], and the most common carbonate in soils is calcite (Ming 2006). Further, some of the carbonate minerals in urban soils may also be a product of concrete/cement-derived material deposited in the soil as a consequence of demolition activity (Renforth et al., 2011). In the following section, SIC dynamics in soils is discussed, along with factors and processes that can be managed to enhance urban SIC stocks.

The natural process of pedogenic carbonate formation in soils may be the result of weathering/dissolution of carbonate parent materials through reaction with carbonic acid followed by pedogenic carbonate precipitation of weathering products especially at deeper soil depths (Wu et al., 2009). This process does not contribute to C sequestration as one mole of CO_2 spent in the formation of carbonic acid is released during precipitation of $CaCO_3$. Pedogenic carbonates may also be formed following the weathering of soluble Ca/Mg-bearing silicates (Berner 2004). This reaction consumes two mole of CO_2:

$2CO_2 + 3H_2O + CaSiO_3 \rightarrow Ca^{2+} + 2HCO_3^- + H_4SiO_4$ (Equation 1)

The subsequent calcite precipitation results in C sequestration as only one mole of CO_2 is released:

$Ca^{2+} + 2HCO_3^- \rightarrow CaCO_3 + CO_2 + H_2O$ (Equation 2)

The inorganic C in the precipitated pedogenic carbonate originates mainly from CO_2 generated through the respiration of plant roots by way of photosynthesis and root-associated microorganisms (Cerling 1984). Thus, SIC stocks in urban soils may be enhanced by (i) promoting a productive vegetation cover with an extensive surficial root system releasing large amounts of CO_2 into the soil that results in carbonate dissolution in soil surface horizons. Convection of CO_2, and carbonate and bicarbonate ions with percolating water to deeper soil horizons may subsequently contribute to pedogenic carbonate formation in the subsoil. Thus, another recommendation is (ii) creating and maintaining a soil structure favorable to water percolation from surface to subsurface horizons (e.g., continuous system of coarse pores for water movement into the subsoil). Further, (iii) creating acidic surface soil horizons and alkaline subsoil horizons is another recommendation as carbonates are dissolved under acidic conditions and precipitate under alkaline conditions.

The Ca and Mg contents of the pedogenic carbonates may originate from soil minerals but also from OM decomposition (Ming 2006). Thus, other recommendations include the (iv) addition of Ca- and Mg-rich OM to urban soils, and (v) cultivation of plants producing high amounts of Ca- and Mg-rich plant litter. Further, (vi) addition of Ca and Mg with fertilizer, liming material (i.e., containing calcite or dolomite) or other soil amendments may result in an increase in urban SIC stock by formation of pedogenic carbonates. The (vii) irrigation of urban soils with alkaline water may also result in an increase in SIC stocks (Xu and Liu, 2013; Xu et al., 2012). Further, (viii) adding amendments or conditioners to urban soils that conserve water in the soil column potentially enhance pedogenic carbonate formation (Lal and Follett, 2009). Waste materials in urban soils derived from construction materials may contain Ca- and Mg-silicate minerals. Naturally, these minerals occur in basic igneous rocks including those quarried for aggregate (Renforth et al., 2011). The Ca- and Mg-silicate minerals may also be produced in cement, and iron- and steel-making slag. Weathering of these materials (i.e., Ca-rich minerals such as silicates, hydroxides, sulphates) in urban soils may be subsequently followed by pedogenic carbonate precipitation. Materials include Ca-rich components such as artificial mortars, plaster, concrete, natural basic rock aggregates and slag (Renforth et al., 2009). Using these materials as soil amendments in urban soils is, thus, (ix) another recommendation to enhance SIC stocks in urban soils.

In conclusion, best management practices to enhance SIC stocks under existing urban vegetation cover includes optimizing NPP by fertilization (i.e., applying NH_4^+-producing fertilizers) and irrigation (i.e., with alkaline water). Establishing a new vegetation cover on bare urban soil after construction activities offers several opportunities for enhancing SIC stocks. For example, planting should either be done in fertile urban soil or soil fertility should be improved during site establishment by adding (i) Ca- and Mg-rich OM, (ii) inorganic fertilizers containing Ca and Mg, and (iii) waste materials containing Ca- and Mg-silicate minerals. Plant species with an extensive surficial root system producing high amounts of Ca- and Mg-rich litter should be established. Before planting, soil structure should be improved by deep tillage to reduce soil compaction. However, soil disturbance should be reduced after plant establishment to allow the development of well-structured soil and also the proliferation of deep burrowing earthworms. While surface soil can be acidified by applying NH_4^+-producing fertilizers, burial of alkaline compounds at deeper depths assists in creating alkaline conditions for carbonate precipitation in urban subsoil.

Enhancing organic carbon stocks of urban soils

The persistence of SOC is largely due to complex interactions between SOC and its environment, such as the interdependence of compound chemistry, reactive mineral surfaces, climate, water availability, soil acidity, soil redox state and the presence of potential degraders in the immediate microenvironment (Schmidt et al., 2011). Urban soil management may be directed to enhance SOC persistence by manipulating these interactions. However, an improved understanding of the potential for management targeted specifically at SOC is needed (Scharlemann et al., 2014). Increasing the urban SOC storage for climate change mitigation may include: (i) increasing the rate of input of OM; (ii) decreasing the rate of its decomposition by biological or chemical means; (iii) increasing the rate of its stabilization by physicochemical protection within aggregates and organo-mineral complexes; and (iv) increasing the depth or total soil volume sequestering C at maximum rate (Whitmore et al. 2013). The SOC enhancing management practices may be used to restore SOC in bare urban soils by re-vegetation, and in green space soils by fertilizing, irrigation, reduced soil disturbance and residue management (i.e., returning grass clippings). During construction activities, urban soils can potentially be improved towards organic C accumulation by adding organic amendments such as biosolids, yard and food waste (Brown et al., 2012). In particular, lawn SOC stocks may benefit from management targeted at SOC as lawns are often intensively managed. Further, garden management practices such as the addition of peat, composts, and mulches, and cultivation of trees and shrubs may contribute to greater SOC stocks in urban gardens (Edmondson et al., 2014). However, tree planting and management of existing tree cover likely provides the greatest scope for enhancing urban SOC stocks (Renforth et al., 2011). Urban soils covered by trees are often relatively undisturbed compared to other vegetated urban soils allowing long time for SOC accumulation. However, how to increase C restitution from tree biomass to C storage in urban soils is less well known than the C storage potential of urban trees (Scharenbroch, 2012). Some specific examples of how to enhance urban SOC stocks in soils beneath impervious surfaces and those of unsealed soils are given in the following section.

Soils beneath impervious surfaces

The knowledge about SOC transformation in impervious-covered urban soils is limited (Wei et al., 2014). However, some studies indicate that soil microbial activities under impervious surfaces are reduced (Piotrowska-Dlugosz and Charzyński 2014; Wei et al. 2013). Thus, OM buried in impervious-covered soil profiles may decompose at a lower rate than OM buried in unsealed urban soil profiles. The burial of large amounts of OM during impervious-covered urban soil construction may, therefore, be a management practice to enhance urban SOC stocks. The SOC pool of impervious-covered soils can be further strengthened by adding 'recalcitrant' compounds as those may be the dominant SOC fraction under impervious cover (Raciti et al., 2012a). 'Recalcitrant', in particular, are combustion-derived C compounds as those may be the only non-mineral-associated SOC component aside from fossil C that may be persistent in soil (Marschner et al., 2008). Certain ranges in the combustion continuum, i.e., carbonaceous substance of pyrogenic origin resistant to thermal or chemical degradation by applying specific methods (Hammes and Abiven 2013), are termed 'black carbon (BC)'. Adding BC with charred organic material, charcoal, biochar or soot during the construction of impervious-covered urban soils may particularly enhance the SOC stock. For example, studies by Beesley and Dickinson (2011), Ghosh et al. (2012) and Scharenbroch et al. (2013) indicated that adding biochar or char products to urban soils enhances SOC stocks and improves soil quality.

Asphalt, coal, coal ash and tar can also contribute to urban soil BC (Rawlins et al., 2008; Stahr et al., 2003; Yang et al., 2010). However, research into the use of BC and, especially, biochar to enhance SOC and improve soil quality is in its early stages (Lorenz and Lal, 2014). Specifically, little thought has been given to biochar application to urban soils and gardens (Renforth et al., 2011). Most importantly, BC including biochar cannot be generally regarded as being persistent in soil as many factors influence its environmental fate. Important factors include: (i) soil type including mineralogy (i.e., mineral fraction able to sorb BC into the SOC pool), (ii) combustion conditions (i.e., amount of fuel or biomass, fire frequency and severity), (iii) contents of aromatic precursors in BC, (iv) removal of BC from the soil surface by mixing, (v) climate, (vi) biota (i.e., microbial communities capable of degrading aromatic C), (vii) soil position in the landscape, and (viii) land use practices (Czimczik and Masiello, 2007). Clearly, more studies are needed on the potential to enhance the urban SOC stock and improve urban soil quality by adding BC during construction of impervious-covered soils.

Unsealed soils

As a guiding principle to enhance urban SOC stocks, urban soils should be vegetated and the vegetation maintained to optimize NPP as this is the primary source of soil C inputs such as plant litter and exudates, and dissolved organic carbon (DOC; Rumpel and Kögel-Knabner, 2011). The SOC-enhancing management practices (e.g., reduced physical disturbance, fertilization, irrigation) should, in particular, optimize root growth and root turnover as SOC may be primarily originating from root-derived C inputs (Rasse et al., 2005). Adding BC during the construction of unsealed urban soils may be an appropriate management practice to enhance the SOC stock similar to impervious-covered soils. Further, burial of urban soil including its SOC stock and/or carbonaceous materials at deeper soil depths can contribute to the delivery and long-term persistence of substantial SOC stocks to depths (Chaopricha and Marín-Spiotta 2014). For example, enrichment of urban soil profiles predominantly with anthropogenic OM up to 1.9-m depth was reported for some soils in Stuttgart, Germany (Stahr et al., 2003). Increasing the bulk density in urban sub-soil horizons without affecting plant root development may aide in enhancing SOC stocks by creating unfavorable conditions for decomposition while at the same time allowing transfer of root-derived C into sub-soil horizons. Additional storage of SOC at deeper soil depths in urban areas may also be possible by cultivating plants with deep and prolific roots systems (Kell, 2012). The strategy for enhancing SOC is to select plant species with enhanced transfer of root-derived C into stable mineral-associated SOC fractions. Thus, selection of specific vegetation in planning and design of new developments may be important for enhancing urban SOC stocks. Enhancing sub soil SOC may also be achieved by creating and maintaining favorable soil conditions for below ground DOC input with percolation water (Schrumpf et al., 2013). As retention of DOC in sub soils may be related to the concentration of poorly crystalline iron and aluminum (hydr)oxides with a high specific surface, another potential SOC enhancing practice is adding those mineral fractions during urban soil construction. Adding clay-sized soil fractions, in particular, may provide preferential spots for SOC accumulation (Vogel et al., 2014). Creating optimum conditions for organic C accumulation by promoting root development and decomposer activity such as low bulk density and pH in surface soil horizons is another practice contributing to enhancing SOC stocks. In summary, vegetation species, soil compaction and sward/litter management must be controlled for optimum SOC accumulation in unsealed urban soils (Renforth et al., 2011).

In conclusion, best management practices to enhance SOC stocks under existing vegetation cover are to some extent similar to those for enhancing SIC stocks. Thus, co-benefits for both stocks exist thru optimizing NPP by fertilization and irrigation (especially optimizing belowground NPP for enhancing SOC stocks), adding OM (including BC for enhancing SOC stocks), and reducing soil disturbance. Bare soils should be re-vegetated, in particular, with plants developing deep and prolific root systems to enhance SOC stocks. However, creating acidic conditions at the soil surface for enhancing SIC stocks can reduce NPP of some acid-sensitive plants and, thus, conflict with approaches for enhancing SOC stocks. SOC stocks may also be reduced by deep tillage during site establishment for reducing soil compaction to enhance SIC stocks as decomposition of OM may be promoted. Further, adding ‘recalcitrant’ OM such as BC may enhance SOC stocks but less Ca and Mg may be released during decomposition of ‘recalcitrant’ OM reducing its potential to contribute to enhanced SIC stock formation. Otherwise, adding Ca and Mg with soil amendments for enhancing SIC stocks may also enhance SOC stocks as Ca^{2+} and Mg^{2+} ions facilitate sorptive SOC stabilization by mineral-organic associations.

Practical and applied approaches to enhance urban soil inorganic and organic carbon stocks

The knowledge about applied and practical approaches aimed at enhancing urban SIC and SOC stocks need to be disseminated among stakeholders (e.g., urban planners and land users, local professionals and governments, NGOs, policymakers). However, current understanding of urban soils and their functions is limited (Yigini et al. 2012). Specifically, soil is not a well-developed theme among urban planners and too many knowledge issues exist for them. Helpful may be showing the value of soil functions down to the numbers. Urban planning is currently responding but should lead with regard to urban soils and their functions. Thus, urban land use planning should also consider urban soils and their ability to store SIC and SOC, and direct necessary land-use conversions to soils of lower quality, i.e., those storing less SIC and SOC. Integration of urban development policies is crucial at all levels and participative tools are essential to involve stakeholders and create ownership (Yigini et al. 2012). Soil is forgotten in urban policy legislation. To address this policy gap one possibility may be to highlight the impact of urbanization of soils on food production. Integrated holistic approaches are needed within the air-land-water nexus. To enhance economic soil use, calculating the financial aside the environmental burden of lost and reduced soil functions by urban conversion may be appropriate. To raise awareness urban soil functions should be connected with something people care, e.g., food, water. Awareness should be raised, in particular, within the urban planning process (Yigini et al. 2012). In summary, it is crucial to connect soil functions related to SIC and SOC with (i) water (infiltration, climate regulation) and its economic value, (ii) green spaces (trees, birds, life quality), (iii) history, (iv) landscape, (v) biodiversity, and (vi) quality (eco-service functions, biodiversity, sealing level, drainage ability, contamination ground and air, level of emissions from land, soil map indicators).

Conclusion

Urban population is increasing, and with the expansion of urban areas also increasing is the proportion of urban soils relative to those of non-urban soil resources. Thus, more and more people depend on critically important urban soil-derived ecosystem services such as C storage/regulation for climate change mitigation, nutrient cycling, primary production, flood regulation, pollutant attenuation, regulation of biodiversity, cultural services, and carrying structure and piped utilities. The terrestrial C loss by urban expansion is not negligible compared to those by other land use changes. However, properly managed urban soils can maintain soil C stocks (e.g., by minimizing losses form erosion and decomposition). Further, SIC stocks may be enhanced by mineral carbonation through addition of Ca- and/or Mg-bearing demolition material while SOC stocks may be enhanced by cultivating SOC-accumulating vegetation, and adding OM and, particularly, BC to subsoil layers during soil construction activities. However, the database for urban soil C stocks and its dynamics, especially for many rapidly urbanizing regions is small. Also, long-term field studies are needed to evaluate approaches for intentionally enhancing the urban soil C stocks. Most importantly, awareness about the importance of urban soil C stocks for climate change mitigation and, specifically, for soil-derived ecosystem services must be raised among stakeholders. The improved understanding of processes will facilitate urban planning for enhancing the soil C stocks and the attendant resilience of urban ecosystems, climate change adaptation and mitigation, and human well-being.

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Table 1

Inorganic carbon stocks of urban soils (simplified land use and land cover classification according to Ellis and Ramankutty, 2008)

Depth (cm)	Inorganic carbon (Mg C ha^{-1})	Land use	Land cover	City/Region	Reference
0-10	0.45-0.62	Builtup/ornamental	Herbaceous/trees	Phoenix, AZ, U.S.A.	Kaye et al. (2008)
0-15	1.3	Builtup/ornamental	Herbaceous/trees	Fort Collins, CO, U.S.A.	Kaye et al. (2005)
	0.9; 1.7	Builtup/forestry	Herbaceous/trees	Louisville, KY, U.S.A.	Trammell and Carreiro (2012)
0-20	0-49.5	Builtup/ornamental	Bare/herbacous/trees	Jiangsu Province, China	Xu and Liu (2013)
	1.3-30.7	Builtup/ornamental	Bare/herbacous/trees	Shanghai, China	Xu et al. (2012)
10-30	0.98-1.04	Builtup/ornamental	Herbaceous/trees	Phoenix, AZ, U.S.A.	Kaye et al. (2008)
15-30	2.0	Builtup/ornamental	Herbaceous/trees	Fort Collins, CO, U.S.A.	Kaye et al. (2005)
0-30	12-82	Builtup/crop/ornamental	Bare/herbacous/trees	Stuttgart, Germany	Stahr et al. (2003)
0-60	41.5; 44.4	Builtup/ornamental	Herbaceous/trees	Chicago, IL, U.S.A:	Jo and McPherson (1995)
30-100	67-266	Builtup/crop/ornamental	Bare/herbacous/trees	Stuttgart, Germany	Stahr et al. (2003)
160-180	0.9-21.3	Builtup/ornamental	Bare/herbacous/trees	Shanghai, China	Xu et al. (2012)
*Profile	300	Builtup (brownfield)	Herbaceous	Newcastle upon Tyne, U.K.	Renforth et al. (2009)

*Soil depth not specified

Table 2

Organic carbon stocks of urban soils (simplified land use and land cover classification according to Ellis and Ramankutty, 2008)

Depth (cm)	Organic carbon ($Mg\ C\ ha^{-1}$)	Land use	Land cover	City/Region	Reference
0-10	4.2; 6.4	Ornamental	Herbaceous	Victoria, Australia	Livesley et al. (2010)
	19.1-33.7	Ornamental	Herbaceous/trees	Kaifeng city, China	Sun et al. (2010)
	28.1-70.7; 49.8-68.6	Builtup/ornamental	Herbaceous/trees	Moscow; Serebryanye Prudy, Russia	Vasenev et al. (2013)
	5.0-11.0	Builtup/ornamental	Herbaceous/trees	Phoenix, AZ, U.S.A.	Kaye et al. (2008)
	42.1-44.4	Ornamental	Herbaceous/trees	Richmond, VA, U.S.A.	Gough and Elliott (2012)
	34-44	Builtup/forestry/ornamental	Herbaceous/trees	Boston, MA, U.S.A.	Raciti et al. (2012b)
0-15	12.6-48.9	Ornamental	Herbaceous	Hong Kong, China	Kong et al. (2014)
	23.7; 32.5	Ornamental	Herbaceous; herbaceous/trees	Auburn, AL, U.S.A.	Huyler et al. (2013;2014)
	56.7	Ornamental	Herbaceous/trees	New York City, NY, U.S.A.	Raciti et al. (2012a)

	16.5-80.5	Ornamental	Herbaceous/trees	City of Bloomington, IN, U.S.A.	Schmitt-Harsh et al. (2013)
	30.3-80.5	Builtup/forestry	Herbaceous/trees	Louisville, KY, U.S.A.	Trammell et al. (2011)
	34.9-125.5	Ornamental	Herbaceous	Sixteen cities, U.S.A.	Selhorst and Lal (2012)
0-20	12.8-156.5	Builtup/ornamental	Herbaceous/trees	Chongqing Municipality, China	Liu et al. (2013)
	14.7-110.6	Builtup/ornamental	Bare/herbaceous/trees	Shanghai, China	Xu et al. (2012)
	12; 35	Ornamental; athletic fields	Herbaceous	Irvine, CA, U.S.A.	Townsend-Small and Czimczik (2010)
0-21	86-135	Ornamental	Herbaceous/trees	Leicester, U.K.	Edmondson et al. (2014)
0-25	34.5; 96.5	Builtup/ornamental; ornamental	Herbaceous/trees	Vancouver, B.C., Canada	Kellett et al. (2013)
	119.2; 137.8	Crops; ornamental	Herbaceous	Revda, Russia	Meshcheryakov et al. (2005)
10-25	2.5; 3.9	Ornamental	Herbaceous	Victoria, Australia	Livesley et al. (2010)
0-30	91.7; 174.4	Ornamental; Builtup/ornamental	Herbaceous/trees;	Tianjin Binhai New	Hao et al. (2013)

			Bare/herbaceous/trees	Area	
	5.2; 48.5	Ornamental	Bare/herbaceous; herbaceous/trees	Village landscapes, China	Jiao et al. (2010)
10-30	5.3-7.3	Builtup/ornamental	Herbaceous/trees	Phoenix, AZ, U.S.A.	Kaye et al. (2008)
15-30	9.8; 10.3	Ornamental	Herbaceous; herbaceous/trees	Auburn, AL, U.S.A.	Huyler et al. (2013; 2014)
0-50	44.8; 79.1	Forestry; ornamental	Trees; herbaceous/trees	Revda, Russia	Meshcheryakov et al. (2005)
30-50	5.1; 7.4	Ornamental	Herbaceous; herbaceous/trees	Auburn, AL, U.S.A.	Huyler et al. (2013; 2014)
0-90	107; 159	Ornamental; forestry	Herbaceous/trees	Apalachicola, FL, U.S.A.	Nagy et al. (2013)
0-100	47.5; 184.6	Forestry/ornamental	Herbaceous/trees	Xinjiang, Northwest China; Hubei, Central-south China	Zhao et al. (2013)
	63.9-110.2	Ornamental	Herbaceous/trees	Kaifeng city, China	Sun et al. (2010)
	161; 199	Ornamental - non-residential; ornamental - residential	Herbaceous/trees	Leicester, U.K.	Edmondson et al. (2012)
	105-121; 110	Forestry, ornamental; Ornamental	Herbaceous/trees; herbaceous	Baltimore, MD, U.S.A.	Pouyat et al. (2009)
10-150	75.6-818.8; 850.0-	Builtup/ornamental	Herbaceous/trees	Moscow;	Vasenev et al.

	1077.5			Serebryanye Prudy, Russia	(2013)
160-180	7.3-42.9	Builtup/ornamental	Bare/herbaceous/trees	Shanghai, China	Xu et al. (2012)

Figure captions

Fig.1 Optical (left) vs. radar satellite (right) images of the area around Cologne and Bonn, Germany, in unprecedented resolution indicating that the settlement area may be much larger than currently assumed (Credits: USGS/processing by DLR, DLR; http://www.dlr.de/eoc/en/desktopdefault.aspx/tabid-9630/#gallery/24122)

Fig. 2 Large inorganic and organic carbon density (Mg C ha^{-1}) at deeper depths in Shanghai, China (Xu et al., 2012).

Fig. 3 Depth distribution of soil organic carbon density (Mg C ha^{-1}) for industrial, residential and recreational land uses in Moscow (left) and Serebryanye Prudy (right), Russia (Vasenev et al., 2013).